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Session III:

Radon Measurement Methods

Time Series Linear Regression of Half-Hourly Radon Levels in a Residence

David A. Hull
Center for Energy and Environmental Studies
Princeton University
Princeton, NJ 08544-5263

ABSTRACT

This paper uses time series linear regression modelling to assess the impact of temperature and pressure differences on the radon measured in the basement and in the basement drain of a research house in the Princeton area of New Jersey. The models examine half-hour averages of several climate and house parameters for several periods of up to 11 days. The drain radon concentrations follow a strong diurnal pattern that shifts 12 hours in phase between the summer and the fall seasons. This shift can be linked both to the change in temperature differences between seasons and to an experiment which involved sealing the connection between the drain and the basement. We have found that both the basement and the drain radon concentrations are correlated to basement-outdoor and soil-outdoor temperature differences (the coefficient of determination varies between 0.6 and 0.8). The statistical models for the summer periods clearly describe a physical system where the basement drain pumps radon in during the night and sucks radon out during the day.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.[1]

1. INTRODUCTION

Regression modelling of half-hourly averaged radon data is generally difficult for several major reasons. Radon levels are dependent on a large number of environmental and house-specific factors, many of which are not measured on a continuous basis. Despite the fact that important parameters like temperature and pressure are regularly measured, they often vary widely from location to location even within or beneath the same structure. In addition, there is often a time lag between changes in forcing parameters and the related impact on radon levels. This lag is both natural and a result of instrumentation and measurement techniques. Unfortunately, even

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the length of the lag often varies depending on the season or even with the time of day. Finally, the magnitude of the influence of the various parameters on radon concentration does not remain constant from day to day. While these variations may not visually cloud the patterns in the data, they create very poor regression models. This highlights the danger of "blind" regression modelling. Apparent strong patterns in the data may often lead to poor regression fits for the reasons mentioned above. In fact, a very poor fit over a small fraction of the data can appear to invalidate the entire model.

2. APPROACH, HOUSE DESCRIPTION, AND LIST OF MEASURED PARAMETERS

Despite the problems mentioned above, it is still possible to pick out periods during which the influence of certain parameters is particularly obvious. These periods would not only be easier to model, but the appropriate physical processes would be equally more obvious. In particular, during the summer and fall of 1988, the basement and subslab radon levels in Research House 22 (H22) exhibit strong diurnal patterns that seem to be linked to changes in temperature and pressure.

H22 is an old balloon-construction three story home with a full basement. It has radiative heating and zoned cooling, with the particular advantage that the air-conditioning unit is on the third floor, thereby minimizing its impact on basement and subslab radon levels. The basement has three drains: two communicate in the main section (east and west) which have identical radon levels, and a third in the crawlspace that is probably blocked. The drain radon level referred to in the rest of this paper is measured in the west drain. The data are divided into nine periods that range from 7 to 11 days in length and are spread out over the year between late May and early October. The period length is a compromise between using short periods which produce better fits and long periods which provide a much more comprehensive model. The parameters that are used in this study are: basement radon, subslab radon (in this case taken to be the radon level in a basement drain), outdoor temperature, basement temperature, soil temperature, and the pressure difference between the basement and the outdoors, all measured continuously and recorded as half-hour averages. The pressure difference between basement and subslab was also measured, but its total magnitude was inside the range of instrument error and so it was excluded. A complete list of the parameters and their abbreviations is given in Table 1. The goal of this analysis will be to explain the half-hourly patterns in the radon variables using linear regression models of temperature and pressure differences.

3. A FIRST LOOK AT DATA

The radon levels in the basement average 20-50 pCi/L, while the average in the drain ranges between 3000 and 5000 pCi/L. The extraordinarily high levels in the drain led to the assumption that this provided the primary means of radon entry into the basement. So, on Julian day 211, the east and west drains were blocked to test the amount of communication between the basement and the drain. This significantly reduced the radon in the basement for about 10 days. Subsequently, the radon concentration climbed back up to its original level. This experiment reinforced the hypothesis that blocking the path of least resistance does not eliminate the radon

problem, it only causes the radon to find a different path of flow. Although the average radon concentration remained the same, the diurnal patterns were significantly altered. Unfortunately, this also corresponded to the change in seasons from a hot summer to a cooler autumn, so it is difficult to distinguish between the two effects. Thus, the radon data that will be studied have two distinct patterns: summer with the drains open and fall with the drains blocked. A brief description of the data periods is given in Table 2, while complete graphs of the different periods are given in Ref. [1].

A quick scan of the data reveals a strong diurnal pattern (Figs. 1-4). During the summer, the basement and drain radon levels seem highly correlated, peaking in the middle of the night and bottoming out during the heat of the day. During Periods 4 and 5, the hottest part of the summer, the levels in both the drain and the basement drop to almost zero during the day. Some of the summer peaks also have an unusual dip right in the center of the drain radon levels (Figs. 1 and 3). The fall patterns show a similar diurnal cycle, but the drain radon now peaks during the hottest part of the day, 12 hours out of phase with the basement radon (Figs. 2 and 3). The drain radon continues to show the double peak structure which occurred during the summer, but now the dips are even larger.

4. MODELLING APPROACH: STRUCTURE AND METHODOLOGY

The patterns described in Section 3 indicate that a model based on the basement-outdoor temperature difference should produce a good fit to the data. In the initial analysis, we will compare the explanatory powers of two variables: the basement-outdoor temperature difference (T_b-T_o) and the basement-outdoor pressure difference ($dP_{b/o}$), using a time series regression model. Previous time series radon models using half-hourly measurements indicated very strong autocorrelations between the current radon level and that lagged by half an hour [2]. Thus, the autoregressive terms tend to overwhelm the natural driving parameters. Using first-difference models eliminated the autocorrelation, but they also eliminated most of the variation in the dependent variable and the resulting models were relatively poor (the Coefficient of Determination, R^2, <0.5). Consequently, in the framework of the present study, we decided to use simple moving average models. The descriptive power of the independent variables will be evaluated by including lags of 0, 1, 2, 3, and 4 hours. Though the data are accurate to the half-hourly level, hourly lags will be employed in order to keep the models simple (i.e., to hold the number of parameters to a minimum). All coefficients that are not individually significant by a t-test (based on a p-value less than 0.05) [3] will be excluded and the overall quality of the model will be assessed using the R^2 statistic. The choice of 4 hours as the maximum lag was based on trial and error modelling of the data which indicated that longer lags did not significantly improve the models. In addition, the models were adjusted so that all the parameters of each variable had the same sign. For example, if basement radon is positively correlated with $dP_{b/o}$ at a lag of 1 hour, it should not be negatively correlated at a lag of 2 hours. While not necessary in an ordinary time series model, this sign correction is important here to maintain the physical significance of the parameters.

The use of lag terms in a moving-average model does not actually mean that the current radon levels are directly dependent on the temperature differences 2 or 3 hours before. It is merely an abstract method of describing the measured radon as consisting of a mixture of old and new air, where the air infiltration is directly dependent on temperature and pressure differences. Thus, a 2-hour temperature lag term reflects the amount of radon entrained by the air that escaped from the soil 2 hours before which is still present at the measurement location.

5. DRAIN RADON MODELS

5.1 Summer Period

The statistical analysis was initially conducted on the data from Period 1 (148-155 Julian), using the drain radon as the independent variable. The models for this period are given in Table 3. The basic temperature and pressure difference models (3a and 3b) show very poor fits (R^2 of 0.366 and 0.381, respectively), which is rather surprising and contrary to what one would expect just by looking at the data. However, it was unusually cool during the early part of the sample and it rained quite heavily on the last day of the period. If these periods are excluded, both the temperature and the pressure models (3c and 3d) improve tremendously (R^2 of 0.816 and 0.669). From these results, it appears that the temperature difference works at least as well as the pressure difference to describe the drain radon variation. In addition, the high R^2 value suggests that the temperature difference represents the primary driving force for radon entry into the drain, which means that the drain must have some connection to outdoor air. (As yet, an actual exit for the drain has not been discovered.)

The small dips in the drain radon concentration that are noticeable in the peaks on days 148-150 can be explained by further refining the model. It was noticed that they correspond to the times when the outdoor temperature drops below the soil temperature. Therefore, the model has been expanded to include the term $[T_s-T_o]^+$, where '+' implies that the negative values of (T_s-T_o) have been set to zero. The new model (3e) provides only a slight improvement on the simple temperature difference model (R^2=0.853); however, when it is reapplied to the entire week (3f), it represents a substantial improvement over the original model, indicating that the poor fits (i.e., low R^2 values) were a result of an inadequate model rather than the environmental conditions.

Before proceeding, it is important to find a physical description of the system that explains the results of the statistical model. Without such backing, the statistical models have very little value. The models imply that there is a three level system consisting of the soil, a pocket of outdoor air that enters through the drain, and the basement air above the drain. In order to maximize the stack effect, and thus the radon flow, there must be a positive temperature gradient through the entire system; i.e., $T_s < T_o < T_b$. Since the model describes a summer period, the soil temperature is almost always cooler than the outdoor temperature. This means that the dominant effect in the model is the basement-outdoor temperature difference. For this reason, we ignore the times when T_s is

less than T_o and restrict the model to the term $[T_s-T_o]^+$. This corresponds to the times when the soil-outdoor temperature difference acts against the main driving force $[T_b-T_o]$. Also, from a statistical standpoint, since both T_b and T_s are nearly constant over the entire week, the differences $[T_b-T_o]$ and $[T_s-T_o]$ would be almost collinear; thus, they should not both be used in the same model. It might seem that, if the soil temperature were less than the outdoor temperature, the radon flow into the drain from the soil should be cut off completely. This may well be the case, but the duration of this period is short enough that this doesn't substantially affect the overall concentration in the drain, and thus we do not see this pattern in the data.

The Continuous Radon Monitor (CRM) measuring the drain radon takes samples from the air a short distance below the basement floor. When the variable $[T_b-T_o]$ is positive, radon levels will tend to increase because air with a very high radon concentration is being pumped upward from deeper in the drain. Similarly, when $[T_b-T_o]$ is negative, radon levels will tend to decrease because cleaner basement air is being sucked down the drain.

Unfortunately, because of instrumentation problems, we have no pressure measurement which is equivalent to $[T_s-T_o]$, so it is impossible to construct a more comprehensive pressure model. Since the current statistical model does have a strong physical justification, we will extend it to the rest of the summer data.

As we mentioned previously, during the period which includes the hottest part of the summer (Julian days 189-204), the radon levels in the drain drop off to almost zero when $[T_b-T_o]$ is negative. This means that a radon model using $[T_b-T_o]^+$ would provide a better fit to the data. In addition, since $[T_s-T_o]^+$ is never significant in this period, it can be left out of the model completely. The temperature model and a similar pressure model for this period are given in Table 4. (In this and in any future tables, the parentheses give the R^2 value for the model with no intercept. However, since intercept and no-intercept R^2 values are not equivalent, the intercept values are provided first to allow comparisons between models.) We note that they provide a very strong fit to the data. A likely physical explanation is that the temperature rises so quickly that, as soon as the outdoors becomes hotter than the basement, a strong airflow out through the drain drops the radon levels to zero.

We can combine the two temperature models given above (3f,4a) to produce a comprehensive model structure for describing the radon concentrations in the drain by dividing $[T_b-T_o]$ into two terms:

$$Rn_{dr} = a_i[T_b-T_o]_i^+ + b_i[T_o-T_b]_i^+ + c_i[T_s-T_o]_i^+ + k; \quad i = 0,\ldots,4 \qquad (1)$$

where i represents the number of hours of the lag and k is the intercept. Splitting up the basement-outdoor temperature difference makes physical sense, since the change in radon concentration from that in the air pumped from deeper in the drain will be different from the air pumped in from the basement. The fits for each summer period using this model are presented in Table 5. While the coefficients and the significant parameters vary widely from period to period, we note that, overall, the model structure provides a relatively accurate description of the radon concentrations in the drain. The only exception is model 5c, which fares very poorly. A likely reason

for this is that the temperatures are much cooler than is typical for the summer, and so these data do not follow the patterns of the summer model. However, if we examine only the first part of the period (days 175-179), the model does fit ($R^2 = 0.676$), although not as well as for the rest of the data [see Table 6(a)]. The reason that the first part of the data fits the model while the second part does not may be that it takes several days of cooler weather for the summer model to deteriorate.

The adequacy of the summer model is further evaluated by comparing measured and modeled radon levels (see Fig. 1). For the most part, the models appear to fit very well, showing no consistent bias. A few exceptions, like day 148 of Period 1, can be excluded and the results of the models improve substantially [see Table 6(b,c)]. The value of this approach is that it allows one to see if a model is uniformly poor or whether it does not fit only a small part of the data. It is clear that the unusually low peak on day 194 results in a systematic under-fitting of all the rest of the days in that period. Another observation is that the modelled radon values drop below zero during Period 2. These can be rounded up to zero to produce an even better fit for those data. Since there is no systematic under-fitting or over-fitting of the peaks during any period, other than the exception pointed out above, a linear model should be sufficient to describe the data. It is possible that higher order models would provide a good fit for the cooler periods as well (perhaps the process is only locally linear); this represents a good direction for future work.

5.2 Fall Period

While the summer period was dominated by the basement-outdoor temperature difference, the fall period, with the drain blocked to limit contact with the basement, produces a dramatically different diurnal pattern of radon flow. Now, the drain radon levels will be dominated by the difference between the outdoor and the soil temperature. When the outdoor air (i.e., the air in the drain) is warmer than the soil, then the radon will tend to flow into the drain. This corresponds to the peaks that appear in the actual data. These peaks are 12 hours out of phase with the ones that occurred during the summer, appearing during the hottest part of the day. Now, the dip in the center of the peak corresponds to the time when the outdoor temperature rises above that of the basement, indicating that, despite the sealing, the basement air still manages to find a path into the drain. Thus, these dips correspond to cleaner and colder basement air sinking into the drain.

The model structure will be similar to the summer model, but now the outdoor-soil temperature difference will dominate. The term $[T_b-T_o]^+$ will be excluded for similar reasons to those given for the summer model above:

$$Rn_{dr} = a_i[T_o-T_s]_i^+ + b_i[T_s-T_o]_i^+ + c_i[T_o-T_b]_i^+ + k; \quad i = 0,\ldots,4 \qquad (2)$$

where i represents the number of hours of the lag and k is the intercept. Each of the fall periods is fit to this model, and the results are presented in Table 7. Obviously, this model does not fit these data nearly as well as the previous one. Despite the fact that Period 6 could not be fit to the model, the same patterns that were mentioned above do appear there as well. The magnitude of the coefficients do not seem to be consistent enough to

produce a coherent model. This may mean that the initial effects of blocking the drain had not yet completely worn off. The comparison plots of measured versus modelled data (see Fig. 2) also show that the fit, although reasonable, is not as accurate as for the summer model.

As was mentioned previously, it is difficult to determine whether the fall model structure is due entirely to the impact of the blocked drains or whether the cooler weather makes most of the difference. In order to test the relationship between the two factors, the fall model was run on some of the cooler summer data (days 179-184 of Period 3, where the summer model didn't fit) when the drains were open (see Table 8, $R^2 = 0.596$). The fit was poorer than for the rest of the fall data, but it did indicate that cooler weather rather than the blocked drains may explain much of the difference in the patterns of radon flow.

5.3 Conclusions

Thus, the drain radon concentrations during the summer and fall in H22 can be adequately modelled (R^2 generally between 0.7 and 0.8) using the temperatures in the basement, the outdoors, and the soil. Although, the coefficients are quite different from week to week, two distinct model structures emerge, one for the summer and one for the fall. While sealing the drain-basement connection serves to accentuate the difference between the seasons, it is not completely clear whether it has a major effect on the drain radon levels.

It should be mentioned that the current model structure is designed to facilitate the physical interpretation of the parameters. Since T_b and T_s are nearly constant over a 10 day measurement period, the terms $[T_b-T_o]$ and $[T_s-T_o]$ differ to first order by a constant and thus are equivalent in terms of regression modelling. Since some of the information is redundant, it would be misleading to fully use both terms in any model. In fact, both the summer and the fall models of the drain radon are actually sub-models of the following general model:

$$Rn_{dr} = a_i[T_b-T_o]_i^+ + b_i[T_o-T_b]_i^+ + c_i[T_o-T_s]_i^+ + d_i[T_s-T_o]_i^+ + k \quad (3)$$

where i ranges from 0 to 4 and represents the hours of lag of the parameter and k is the intercept. In practice, the collinearity of $[T_b-T_o]$ and $[T_s-T_o]$ forces us to exclude one of these terms from each model. The terms chosen for each model are the ones that provide the best physical description.

6. BASEMENT RADON MODELS

Now that the drain radon concentration is well described by temperature and pressure models, we can examine the basement radon. Inspection of Figs. 3 and 4 reveals that it is strongly related to basement-outdoor temperature and pressure differences. It is not surprising that, during the summer period with the drains open, basement radon levels are also highly correlated with the drain radon levels, following the same diurnal cycle. The first set of models will measure the extent to which the variation in basement radon can be explained by Rn_{dr}, $[dP_{b/o}]$, and $[T_b-T_o]$ for Period 1 (148-155 Julian). The results are presented in Table 9. The models show

that the drain radon and the pressure difference alone provide a reasonable fit, with the temperature difference being much poorer. The best model combines the drain radon with the pressure difference and results in an excellent fit ($R^2 = 0.860$). This combined model is necessary because the basement radon does not show the dips associated with soil-outdoor temperature difference, indicating that the drain is not the only means of radon entry into the basement. Therefore, the basement-outdoor pressure difference in the combined model measures radon entry into the basement from sources other than the drain.

Given the success of the previous models, we will expand the analysis to all of the summer periods using the following model structure:

$$Rn_b = a_i[Rn_{dr}]_i + b_i[dP_{b/o}]i + k \qquad i = 0,\ldots,4 \qquad (4)$$

where i is the time lag in hours and k represents the intercept. The results are presented in Table 10, along with simple pressure difference models for comparison. The reason that a simple pressure model is markedly inferior to the model given above is that it ignores the soil-outdoor parameter that is so important in describing the radon levels in the drain. The full temperature difference model used to model drain radon could also be applied here, but since this is represented by the drain radon values, it is more convenient to deal with the simple model.

The basement model provides an excellent fit for almost every period. The only period where the R^2 value drops below 0.75 is when the drain radon is very poorly modelled, during a stretch of cooler weather. The measured versus modelled plots (see Fig. 3) also show the quality of the fit. In fact, if just a few days are excluded (say, days 150, 166, 168, 188, and 198 Julian), it is likely that the R^2 would exceed 0.95. It should be mentioned that during many periods, particularly the later part of the summer when the soil temperature has very little impact, the drain radon concentration and the basement-outdoor pressure difference are highly correlated. In terms of the models, this means that two factors are likely to be confounded so that the actual values of the coefficients are unreliable, although the overall model structure is still accurate.

The fall period simplifies the basement model even further. Since the drains are plugged, the basement radon no longer shows the strong dependence on the subslab radon concentration. (Although they now have a reasonable negative correlation, this is basically incidental.) Therefore, the fall radon levels in the basement will now be modelled only by temperature and pressure differences. The results are presented in Table 11. For the fall periods, the simple pressure models have a much better fit than they did in the summer. This indicates that, when the drain is blocked, the basement-outdoor pressure difference represents the driving force for radon entry into the basement. Even the temperature models produce better fits than before. Plots of the measured versus modelled values are given in Fig. 4.

7. SIMPLIFIED MODELS

In general, models are designed to explain the maximum amount of variance in the data. While this method produces the most accurate model, it does not necessarily produce a model that is easy to interpret. In

particular, the radon models developed previously are extremely difficult to analyze because the significant time series parameters vary widely from week to week. We can simplify the radon models considerably and hopefully aid in their interpretation by restricting each parameter to a single lag. In order to minimize the information loss from this step, we will now allow for half-hour lags. The new simplified drain models are given in Table 12 (for convenience the brackets give the R^2 of the original model) while the basement models are presented in Table 13. The optimum single lag term for each parameter was calculated by an exhaustive search of all reasonable combinations for each model. This required a lot of time and effort, one disadvantage of the simplified model. However, a close approximation of the best model is relatively easy to calculate.

While the simplified summer drain radon models look very different as the season progresses, the actual drain radon data remain much the same. The difference lies in the flat troughs of the daily cycle that appear in the later periods. They result from the simple fact that radon concentrations must be greater than zero. The term $[T_o-T_b]^+$ reflects the airflow from the basement into the drain, and once the drain radon falls to zero, as it rapidly does during the late summer, the term has no impact on the radon level and thus it disappears. Similarly, the term $[T_s-T_o]^+$ has no effect in Period 4 because the outdoor temperature rarely drops below the soil temperature, and even when it does it is only by a degree or two.

The simplified fall radon models all look remarkably similar, with all terms having a 2-hour lag. This indicates that the fall drain model should be able to fit all of the data at once. The comprehensive fall model is presented in Table 14, in both the complex and the simplified form. Thus, it seems that during the fall, the drain radon follows a relatively consistent pattern dominated by the outdoor-soil temperature difference. Note that, although the term $[T_o-T_b]^+$ has the largest coefficient, T_o is so rarely higher than T_b that this parameter is only of secondary importance.

The new basement radon models are not nearly as consistent as the fall drain radon models described previously. In the summer models, the drain radon term remains relatively consistent but the pressure difference term varies considerably both in time lag and in magnitude. The fall models are somewhat better, with the magnitude of the pressure coefficient varying between 30 and 40 pCi/L/Pa.

8. CONCLUSIONS

One very significant element of this analysis is missing: a study of the residuals. It is excluded since the goal of this study was not to produce an ideal time series model of radon behavior, but to come up with rougher but simpler relations that lend themselves to physical interpretation. It is important not to get overwhelmed by statistics to the extent that one loses sight of the final goal. Up to this point, good time series models of short-term radon behavior are almost non-existent. This has a lot to do with the complexity and variability of radon measurements. In fact, the residuals of the radon models in this study show a strong auto-correlation, because the magnitude of the temperature effect varies considerably, even on a day to day basis. Although this means that the models have significant problems, it does not invalidate them as guides to a

physical analysis of the system.

The models developed in this analysis show that temperature differences can explain between 60 and 80% of the variation in radon concentration in the basement and the drain of House 22. They reveal a relatively complex system of air dynamics where the drain works as a pump, pulling radon into the basement during one phase of a diurnal cycle and sucking radon out during the other phase. As the temperature differences between the soil, the basement, and the outdoors change as summer shifts into fall, an entirely new pattern of drain radon flow emerges. The drain sealing probably has some impact on the dynamics of the system, although its precise effect cannot properly be measured. Although the models reveal general patterns, their complexity makes them quite difficult to analyze. The simplified models sacrifice between 5 and 10% in terms of the coefficient of variance, but they prove to be much easier both to compare and to interpret.

Because of the quantity of data, particularly when examined at the half-hour level, this study has been limited to two seasons out of the year. An expansion into both the winter and the spring seasons would be desirable. Unfortunately, the clear diurnal pattern that is present in the summer and fall data completely disappears during the winter, making modelling much more difficult. In wintertime, T_o never rises above T_b, meaning that the drain does not pump radon out of the basement and the levels remain relatively high. However, the winter season certainly deserves further study. Despite the pitfalls, time series regression analysis can produce reasonable models for half-hour radon data.

9. ACKNOWLEDGEMENTS

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Table 1 - Parameters measured continuously in H22

Rn_b - basement radon measured by a Wrenn chamber (pCi/L)
Rn_{dr} - radon level in drain beneath the basement measured by a CRM (pCi/L)
T_b - basement temperature (^oC)
T_o - outdoor temperature (^oC)
T_s - soil temperature (^oC)
$dP_{b/o}$ - pressure difference, ambient - basement (Pa)

Table 2 - Description of the data periods

Period	Julian days	No. days	Reason for exclusion
1	148-155	7	
-	155-164	10	cooler weather / uncharacteristic summer pattern
2	165-174	10	
3	175-184	10	
4	185-194	10	
5	195-203	9	
-	204-210	7	heavy rains / probable drain flooding
-	211-220	10	initial attempt to block drains
-	221-232	12	drains sealed / radon levels decreased
6	233-242	10	
7	243-252	10	
8	253-262	10	
9	263-273	11	
-	274+	-	drains re-opened

Table 3 - Drain radon models (Period 1)

Days 148-155

a) $Rn_{dr} = 287[T_b-T_o] + 5895$ $R^2 = 0.366$

b) $Rn_{dr} = 1138[dP_{b/o}] + 915[dP_{b/o}]_{1hr} + 4975$ $R^2 = 0.381$

Days 148.5-154

c) $Rn_{dr} = 248[T_b-T_o] + 249[T_b-T_o]_{2hr} + 6737$ $R^2 = 0.816$

d) $Rn_{dr} = 1084[dP_{b/o}] + 1088[dP_{b/o}]_{2hr} + 5114$ $R^2 = 0.669$

e) $Rn_{dr} = 266[T_b-T_o] + 312[T_b-T_o]_{2hr} - 400[T_s-T_o]^+_{1hr} - 301[T_s-T_o]^+_{2hr} + 7147$ $R^2 = 0.853$

Days 148-155

f) $Rn_{dr} = 299[T_b-T_o] + 314[T_b-T_o]_{2hr} - 594[T_s-T_o]^+_{1hr} - 623[T_s-T_o]^+_{3hr} + 7242$ $R^2 = 0.764$

Table 4 - Drain radon models (Days 189-204)

a) $Rn_{dr} = 793[T_b-T_o]^+ + 562[T_b-T_o]^+_{2hr}$ $R^2 = 0.783$ (0.860)

b) $Rn_{dr} = 2256[dP_{b/o}]^+ + 3219[dP_{b/o}]^+_{1hr} + 3310[dP_{b/o}]^+_{2hr}$

$R^2 = 0.836$ (0.894)

Table 5 - Drain radon revised model (all summer periods)

Period	Model
a) 148-155	$Rn_{dr} = 556[T_b-T_o]^+ + 590[T_b-T_o]^+_{2hr} - 152[T_o-T_b]^+ - 209[T_o-T_b]^+_{2hr} - 646[T_s-T_o]^+ - 936[T_s-T_o]^+_{2hr} - 276[T_s-T_o]^+_{4hr} + 5604$ $R^2 = 0.802$
b) 165-175	$Rn_{dr} = 760[T_b-T_o]^+ + 462[T_b-T_o]^+_{1hr} - 132[T_o-T_b]^+ - 213[T_o-T_b]^+_{4hr} - 1131[T_s-T_o]^+ - 871[T_s-T_o]^+_{1hr} + 2395$ $R^2 = 0.827$
c) 175-185	$R^2 = 0.400$ no consistent model structure
d) 185-195	$Rn_{dr} = 1093[T_b-T_o]^+ + 445[T_b-T_o]^+_{2hr} + 222[T_b-T_o]^+_{4hr} - 932[T_s-T_o]^+ - 872[T_s-T_o]^+_{1hr} - 405[T_s-T_o]^+_{2hr} + 202$ $R^2 = 0.756$
e) 195-204	$Rn_{dr} = 856[T_b-T_o]^+ + 465[T_b-T_o]^+_{2hr}$ $R^2 = 0.741$ (0.832)

Table 6 - Drain radon: reduced modelling periods

Period	Model
a) 175-179	$Rn_{dr} = 772[T_b-T_o]^+ + 578[T_b-T_o]^+_{1hr} - 708[T_s-T_o]^+ - 1058[T_s-T_o]^+_{1hr} + 1937$ $R^2 = 0.676$
b) 149-155	$Rn_{dr} = 742[T_b-T_o]^+ + 334[T_b-T_o]^+_{3hr} - 237[T_o-T_b]^+ - 188[T_o-T_b]^+_{4hr} - 685[T_s-T_o]^+ - 582[T_s-T_o]^+_{1hr} - 416[T_s-T_o]^+_{3hr} + 6144$ $R^2 = 0.896$
c) 185-194	$Rn_{dr} = 1143[T_b-T_o]^+ + 502[T_b-T_o]^+_{2hr} + 183[T_b-T_o]^+_{4hr} - 1089[T_s-T_o]^+ - 822[T_s-T_o]^+_{1hr} - 464[T_s-T_o]^+_{2hr} + 376$ $R^2 = 0.823$

Table 7 - Drain radon: fall models

Period	Model
a) 233-243	$R^2 = 0.362$ no consistent model structure
b) 243-253	$Rn_{dr} = 1104[T_o-T_s]^+ + 1490[T_o-T_s]^+_{2hr} + 1069[T_o-T_s]^+_{3hr} + 857[T_o-T_s]^+_{4hr} - 154[T_s-T_o]^+ - 260[T_s-T_o]^+_{3hr} - 1984[T_o-T_b]^+ - 1704[T_o-T_b]^+_{1hr} - 2570[T_o-T_b]^+_{2hr} - 2032[T_o-T_b]^+_{3hr} - 1966[T_o-T_b]^+_{4hr} + 5048$ $R^2 = 0.718$
c) 253-263	$Rn_{dr} = 1271[T_o-T_s]^+_{1hr} + 941[T_o-T_s]^+_{2hr} + 645[T_o-T_s]^+_{3hr} - 307[T_s-T_o]^+_{2hr} - 1835[T_o-T_b]^+_{1hr} - 1537[T_o-T_b]^+_{2hr} - 1063[T_o-T_b]^+_{3hr} - 842[T_o-T_b]^+_{4hr} + 3955$ $R^2 = 0.629$
d) 263-274	$Rn_{dr} = 732[T_o-T_s]^+ + 708[T_o-T_s]^+_{1hr} + 1177[T_o-T_s]^+_{2hr} + 1383[T_o-T_s]^+_{4hr} - 367[T_s-T_o]^+_{1hr} - 170[T_s-T_o]^+_{4hr} - 1101[T_o-T_b]^+ - 1268[T_o-T_b]^+_{1hr} - 2033[T_o-T_b]^+_{2hr} - 2872[T_o-T_b]^+_{4hr} + 5429$ $R^2 = 0.760$

Table 8 - Drain radon: fall model tested on summer data

Period	Model	
179-184	$Rn_{dr} = 201[T_o-T_s]^+_{3hr} - 464[T_s-T_o]^+ - 136[T_s-T_o]^+_{4hr} - 644[T_o-T_b]^+_{1hr} + 6824$	$R^2 = 0.596$

Table 9 - Basement radon models (Days 148-155)

a) $Rn_b = 0.0084[Rn_{dr}]_{1hr} - 4.6$	$R^2 = 0.743$
b) $Rn_b = 11.0[dP_{b/o}]_{1hr} + 14.8[dP_{b/o}]_{2hr} + 32.0$	$R^2 = 0.636$
c) $Rn_b = 3.2[T_b-T_o]_{1hr} + 43.9$	$R^2 = 0.481$
d) $Rn_b = 0.0067[Rn_{dr}]_{1hr} + 1.3[T_b-T_o]_{1hr} + 4.9$	$R^2 = 0.789$
e) $Rn_b = 0.0041[Rn_{dr}]_{1hr} + .0021[Rn_{dr}]_{2hr} + 6.3[dP_{b/o}]_{1hr} + 7.2[dP_{b/o}]_{2hr}$	$R^2 = 0.860$ (0.957)

Table 10 - Basement radon: summer models

Period	Models	
a) 148-155	$Rn_b = 0.0041[Rn_{dr}]_{1hr} + .0021[Rn_{dr}]_{2hr} + 6.3[dP_{b/o}]_{1hr} + 7.2[dP_{b/o}]_{2hr}$	$R^2 = 0.860$ (0.957)
	$Rn_b = 11.0[dP_{b/o}]_{1hr} + 14.8[dP_{b/o}]_{2hr} + 32.0$	$R^2 = 0.636$
b) 165-175	$Rn_b = 0.0025[Rn_{dr}] + .0025[Rn_{dr}]_{2hr} + .0019[Rn_{dr}]_{4hr} + 10.6[dP_{b/o}]_{2hr} + 10.4[dP_{b/o}]_{3hr} + 7.0$	$R^2 = 0.796$
	$Rn_b = 24.7[dP_{b/o}]_{2hr} + 18.2[dP_{b/o}]_{4hr} + 22.6$	$R^2 = 0.660$
c) 175-185	$Rn_b = 0.0021[Rn_{dr}] + .0013[Rn_{dr}]_{2hr} + .0013[Rn_{dr}]_{4hr} + 8.7[dP_{b/o}]_{1hr} + 10.9[dP_{b/o}]_{2hr} + 5.9$	$R^2 = 0.665$
	$Rn_b = 9.8[dP_{b/o}]_{1hr} + 8.2[dP_{b/o}]_{2hr} + 26.5$	$R^2 = 0.461$
d) 185-195	$Rn_b = 0.0047[Rn_{dr}]_{1hr} + .0025[Rn_{dr}]_{4hr} + 8.3[dP_{b/o}]_{1hr} + 7.5[dP_{b/o}]_{3hr} + 5.0$	$R^2 = 0.842$
	$Rn_b = 15.9[dP_{b/o}]_{1hr} + 14.0[dP_{b/o}]_{4hr} + 19.7$	$R^2 = 0.605$
e) 195-204	$Rn_b = 0.0013[Rn_{dr}] + .0028[Rn_{dr}]_{1hr} + .0020[Rn_{dr}]_{3hr} + 3.4[dP_{b/o}]_{2hr} + 2.1$	$R^2 = 0.811$
	$Rn_b = 8.0[dP_{b/o}]_{1hr} + 10.3[dP_{b/o}]_{3hr} + 13.4$	$R^2 = 0.557$

Table 11 - Basement radon: fall models

	Period	Models	
a)	233-243	$Rn_b = 14.5[dP_{b/o}]_{1hr} + 14.8[dP_{b/o}]_{2hr} + 13.7[dP_{b/o}]_{4hr} - 7.3$	$R^2 = 0.779$
		$Rn_b = 3.4[T_b-T_o]^+ + 4.4[T_b-T_o]^+_{4hr}$	$R^2 = 0.725$ (0.875)
b)	243-253	$Rn_b = 14.6[dP_{b/o}]_{1hr} + 16.3[dP_{b/o}]_{2hr} + 5.5$	$R^2 = 0.633$
		$Rn_b = 3.8[T_b-T_o]^+ + 2.9[T_b-T_o]^+_{3hr} + 12.3$	$R^2 = 0.618$
c)	253-263	$Rn_b = 6.0[dP_{b/o}] + 13.2[dP_{b/o}]_{1hr} + 15.9[dP_{b/o}]_{2hr} - 4.9$	$R^2 = 0.754$
		$Rn_b = 2.5[T_b-T_o]^+ + 2.8[T_b-T_o]^+_{1hr} + 16.2$	$R^2 = 0.571$
d)	263-274	$Rn_b = 9.5[dP_{b/o}] + 13.6[dP_{b/o}]_{1hr} + 12.8[dP_{b/o}]_{2hr}$	$R^2 = 0.820$ (0.955)
		$Rn_b = 5.6[T_b-T_o]^+ + 1.6[T_b-T_o]^+_{3hr} + 11.4$	$R^2 = 0.716$

Table 12 - Drain radon: simplified models

	Period	Summer Model	
a)	148-155	$Rn_{dr} = 795[T_b-T_o]^+_{1hr} - 430[T_o-T_b]^+_{1hr} - 1310[T_s-T_o]^+_{2hr} + 6282$	$R^2 = 0.752$ (0.802)
b)	165-175	$Rn_{dr} = 1183[T_b-T_o]^+_{0.5hr} - 210[T_o-T_b]^+_{2hr} - 1747[T_s-T_o]^+_{0.5hr} + 2003$	$R^2 = 0.781$ (0.827)
c)	185-195	$Rn_{dr} = 1493[T_b-T_o]^+_{1hr} - 1925[T_s-T_o]^+_{1hr} + 622$	$R^2 = 0.711$ (0.756)
d)	195-204	$Rn_{dr} = 1253[T_b-T_o]^+_{1hr}$	$R^2 = 0.708$ (0.741)
	Period	**Fall Model**	
e)	243-253	$Rn_{dr} = 2449[T_o-T_s]^+_{2hr} - 485[T_s-T_o]^+_{2hr} - 4938[T_o-T_b]^+_{2hr} + 5818$	$R^2 = 0.648$ (0.718)
f)	253-263	$Rn_{dr} = 1918[T_o-T_s]^+_{2hr} - 338[T_s-T_o]^+_{2hr} - 2975[T_o-T_b]^+_{2hr} + 4198$	$R^2 = 0.574$ (0.629)
g)	243-253	$Rn_{dr} = 2214[T_o-T_s]^+_{2hr} - 564[T_s-T_o]^+_{2hr} - 3570[T_o-T_b]^+_{2hr} + 5808$	$R^2 = 0.668$ (0.760)

Table 13 - Basement radon: simplified models

	Period	Summer Model	
a)	148-155	$Rn_b = 0.0060[Rn_{dr}]_{1hr} + 12.9[dP_{b/o}]_{1.5hr} + 2.6$	$R^2 = 0.864$
		$Rn_b = 23.8[dP_{b/o}]_{1.5hr} + 32.6$	$R^2 = 0.598$
b)	165-175	$Rn_b = 0.0059[Rn_{dr}]_{1.5hr} + 21.3[dP_{b/o}]_{3hr} + 10.3$	$R^2 = 0.775$
		$Rn_b = 39.7[dP_{b/o}]_{3hr} + 23.8$	$R^2 = 0.628$
c)	175-185	$Rn_b = 0.0039[Rn_{dr}]_{1.5hr} + 18.9[dP_{b/o}]_{1.5hr} + 9.9$	$R^2 = 0.641$
		$Rn_b = 17.5[dP_{b/o}]_{1.5hr} + 26.7$	$R^2 = 0.452$
d)	185-195	$Rn_b = 0.0062[Rn_{dr}]_{1.5hr} + 16.0[dP_{b/o}]_{2.5hr} + 7.6$	$R^2 = 0.822$
		$Rn_b = 27.0[dP_{b/o}]_{2.5hr} + 20.4$	$R^2 = 0.588$
e)	195-204	$Rn_b = 0.0051[Rn_{dr}]_{1hr} + 6.1[dP_{b/o}]_{2.5hr} + 4.0$	$R^2 = 0.788$
		$Rn_b = 24.6[dP_{b/o}]_{2.5hr} + 13.4$	$R^2 = 0.537$
	Period	**Fall Model**	
f)	233-243	$Rn_b = 39.5[dP_{b/o}]_{2hr} - 4.1$	$R^2 = 0.751$
		$Rn_b = 6.9[T_b-T_o]^+_{2.5hr} + 3.5$	$R^2 = 0.691$
g)	243-253	$Rn_b = 29.9[dP_{b/o}]_{1.5hr} + 7.2$	$R^2 = 0.620$
		$Rn_b = 6.2[T_b-T_o]^+_{1.5hr} + 15.7$	$R^2 = 0.606$
h)	253-263	$Rn_b = 32.3[dP_{b/o}]_{1.5hr}$	$R^2 = 0.737$ (0.934)
		$Rn_b = 5.2[T_b-T_o]^+_{0.5hr} + 16.8$	$R^2 = 0.567$
i)	263-274	$Rn_b = 35.7[dP_{b/o}]_{1.5hr}$	$R^2 = 0.792$ (0.948)
		$Rn_b = 6.8[T_b-T_o]^+_{0.5hr} + 14.1$	$R^2 = 0.701$

Table 14 - Comprehensive fall drain radon model

Period	Regular Model
233-274	$Rn_{dr} = 579[T_o-T_s]^+ + 751[T_o-T_s]^+_{1hr} + 1026[T_o-T_s]^+_{2hr} + 722[T_o-T_s]^+_{3hr} + 709[T_o-T_s]^+_{4hr} - 388[T_s-T_o]^+_{2hr} - 878[T_o-T_b]^+ - 1547[T_o-T_b]^+_{1hr} - 1669[T_o-T_b]^+_{2hr} - 1214[T_o-T_b]^+_{3hr} - 1841[T_o-T_b]^+_{4hr} + 4655$
	$R^2 = 0.676$

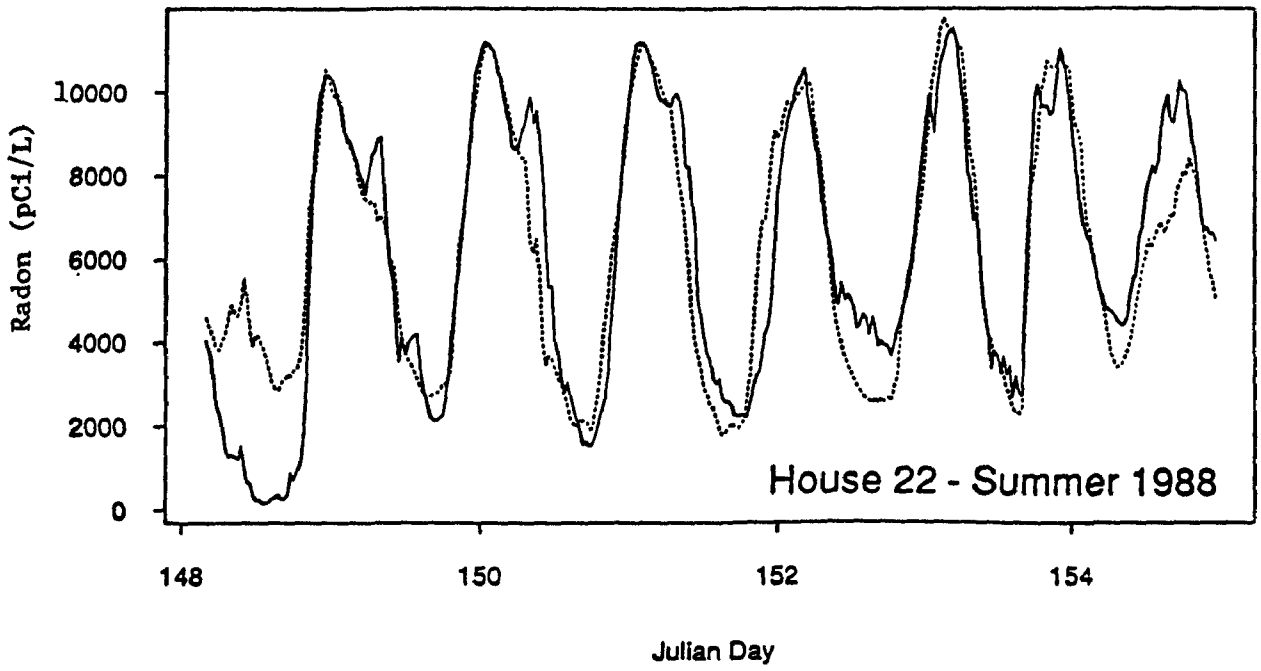


Fig. 1 Comparison of measured drain radon with model predictions using temperature differences (Period 1)

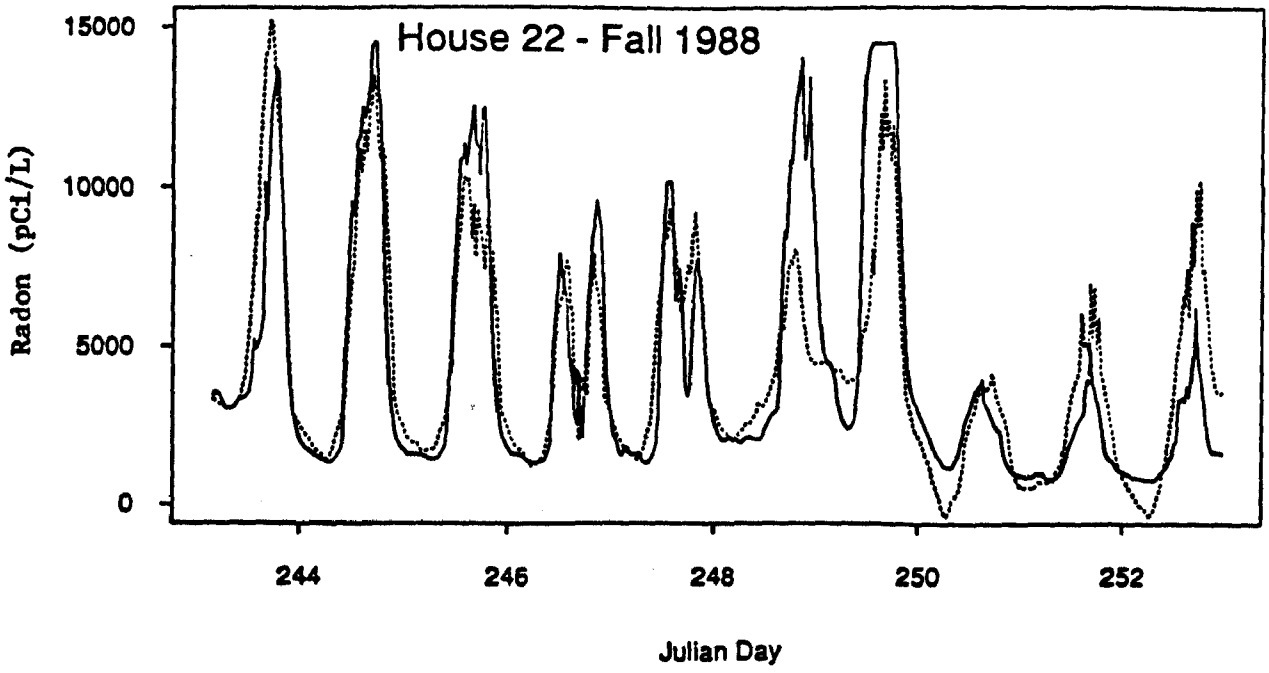


Fig. 2 Comparison of measured drain radon with model predictions using temperature differences (Period 7)

Fig. 3 Comparison of measured basement radon with model predictions using drain radon and basement-outdoor pressure difference (Period 1)

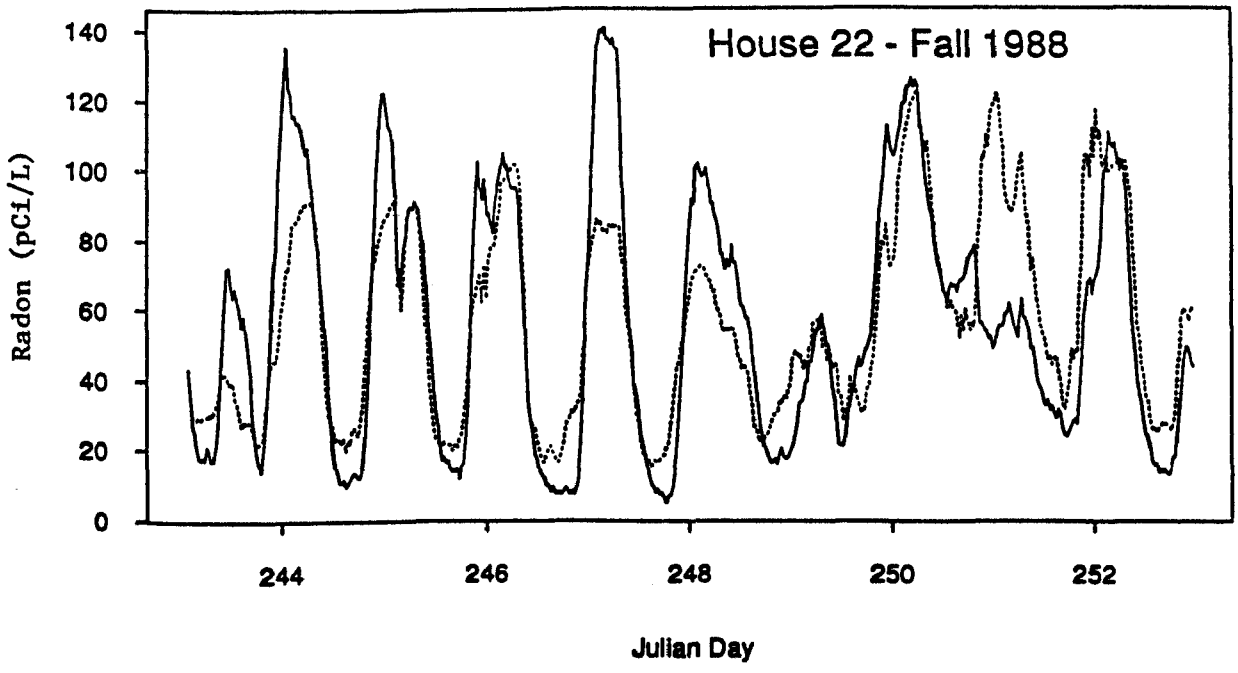


Fig. 4 Comparison of measured basement radon with model predictions using basement-outdoor pressure difference (Period 7)

Experience with the Wire Screen Unattached Fraction Measurement Technique

Van Cleef, Douglas J. and Windham, Sam T.
U.S. Environmental Protection Agency
Eastern Environmental Radiation Facility
Montgomery, Alabama

ABSTRACT

Polonium 218, historically known as Radium A, is commonly considered to be a major contributor to adverse health effects from exposure to radon decay products. The free ions, or unattached fraction, of RaA, by virtue of deeper tissue deposition in the respiratory tract, are believed to be responsible for the greatest portion of this health risk. Because of this, much work has been performed in the development of measurement methods for radioactive particles as small as 0.001 microns, such as unattached Radium A. This work includes development of methods ranging from simple wire screens to complex diffusion batteries. Most of the previous work has been performed using high concentrations of radon or its decay products, providing only theoretical assumptions for the application of the work to environmental levels of radon. This work discusses the application, limitations, and assumptions regarding the use of the single wire screen RaA measurement technique to the lower environmental levels expected in most homes, schools, and office buildings.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

In the past 3 decades, many scientists and engineers have contributed much to the understanding and measurement of unattached Polonium-218 (RaA) particles. Some of the foundational work for the methods evaluated here was performed by THOMAS and HINCHLIFFE[1] and by GEORGE[2] using wire screens as diffusion collection devices. MERCER[3] had previously developed an effective impaction stage diffusion sampler. RAGHAVAYYA and JONES[4] developed a method for estimating RaA unattached fraction (f_a) using wire screens and backup filters. More recent work by VAN DER VOOREN *et al*[5] and RAMAMURTHI and HOPKE [6] have described more sophisticated unattached RaA measurement techniques and limitations of many of the available methods. Some recent work by HOPKE *et al*[7] has included the use of multi-stage wire screen diffusion batteries to enable more complete RaA collection and particle size determination.

The work described in this paper has sought to determine the specific limitations and advantages of the single wire screen collection method, for the following reasons: (1) the method only requires the simultaneous use of two counters, unlike the diffusion batteries or impaction stage devices, which require at least three; (2) the method retains use of the reference filter and its established analysis methods, unlike the wire screen/backup filter combination; and (3) the method is simple and easily adaptable to field work, where the EPA intends to use it.

A secondary goal of this work was to evaluate the use of the reference filter/ wire screen/ backup filter measurement technique for use in field measurement applications.

TESTING FACILITIES

Most of the measurements and analyses were performed at the U.S. EPA's Eastern Environmental Radiation Facility (EERF) in Montgomery, Alabama. The facility has two radon exposure chambers, each with unique environments and sampling limitations. EERF's older chamber, designated Chamber C, is a 3.6 cubic meter volume chamber of wooden construction. It is a single pass, flow through chamber with a heated nichrome wire supplying condensation nuclei (CN). Incoming air is diffused through a perforated particle board upon entry to the chamber, and the radon decay products are typically about 30 minutes old when they reach the sampling points.

The facility's other chamber, designated Chamber A, is a 40 cubic meter volume, recycled environment chamber of mostly aluminum

construction. Condensation nuclei are provided by a wax generator. Radon and its decay products are normally in secular equilibrium in this chamber. Each chamber receives its radon from a series of commercially-available flow-through Ra-226 sources.

SAMPLING DEVICES

Based on the recommendations of GEORGE[2], a 60 mesh wire screen was decided upon as the collection device for unattached RaA. The stainless steel screen was cut to size and pressed into a custom holder of 2 concentric rings, also of stainless steel, to provide a reproducible and convenient method of support for the screen (Figure 1). A commercially available sample head was modified to accept the 52mm diameter screen and a 47mm diameter backup filter, then adapted for use through the chambers' sample ports (Figure 2).

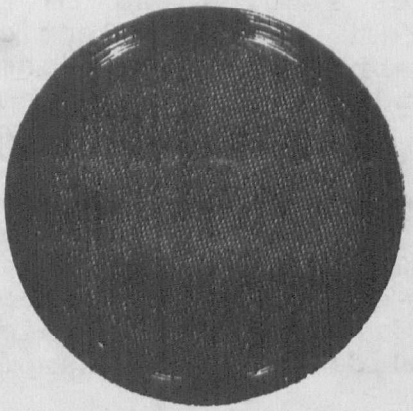


Figure 1. 60 Mesh Stainless Steel Wire Screen

The reference filter sampler is a commercially-available plastic filter holder, normally containing a 25mm membrane-type filter of 0.8 micron pore size. This filter arrangement has been used extensively in radon decay product measurements and serves as the EPA's standard device for such measurements.

SAMPLING AND ANALYSIS

Samples of the chamber atmospheres were taken under a number of widely varied but controlled conditions, in an effort to establish measurement results comparable to those obtained in previous works. Variable sample conditions included sample flow rate, radon and condensation nuclei concentration, and sample duration. All samples taken with the wire screen device were accompanied by simultaneous measurements using the reference filter arrangement. Many samples were taken using the reference filter, wire screen, and backup filter in an effort to estimate f_a from only the reference filter and backup filter values, as described by RAGHAVAYYA and JONES[4]. Sample flow rates varied from 13 to 42 liters per minute, with corresponding screen face velocities of 15 to 48 cm sec^{-1}. Condensation nuclei concentrations were adjusted within a range of about $3x10^3$ to $1x10^5$ particles-cm^{-3}, and chamber radon concentrations varied from under 10 to over 350 pCi-l^{-1}. All condensation nuclei measurements were performed with a single TSI Model 3020 CN counter.

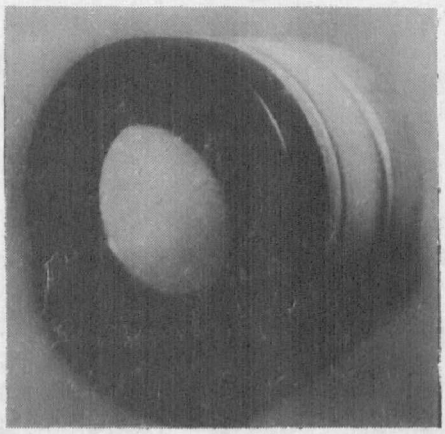


Figure 2. Wire Screen Sample Head

Screen and filter samples were analyzed simultaneously in EERF's radon counting room. The samples were placed particle side down on disks of ZnS(Ag) on a two-inch photomultiplier tube which was enclosed in a light-tight housing. Signal counts were collected by Ludlum model 2200 scalars attached to and controlled by a single personal computer. The computer provided all counter timing and results analysis according to the Thomas-modified Tsivoglou method[8]. Analysis of the reference filter provided individual radon decay product concentrations, but wire screen decay product results had to be modified to correct for collection efficiency and the loss of counts from decay products on the sides or back of the screen mesh. The collection and counting efficiency values were estimated from work by THOMAS and HINCHLIFFE[1] and modified according to actual observations provided by GEORGE[2]. These estimates predict a rapid drop in collection efficiency with changes in face velocity from 0 to 20 cm-sec^{-1}, then a more gradual drop with increasing sample velocities. Collection efficiency estimates, corrected for sample flow rates, were generally between 40 and 50 percent. Calculations included error estimates which were subsequently used to evaluate the validity of the sample results.

RESULTS

As previously stated, data were collected in an effort to reproduce results of previous work with similar equipment, as well as to establish a correlation between theoretical and practical measurement capabilities. One criteria used for comparing our results was to evaluate the shape of the plot of condensation nuclei concentration versus unattached fraction. Several previous works, including those by GEORGE and HINCHLIFFE[9], RAABE[10], and MOHNEN[11] have shown f_a to drop rapidly between 3×10^3 and 1.5×10^4 particles-cm^3. Our ability to reproduce these curves would serve as a good measure of the serviceability of our measurement technique. An additional measure of the value of the method would come from the error analysis of the counting results.

The measurements performed in Chamber A provided results that corresponded very nicely with previous work and theoretical projections, as shown in Figure 3. The curve shape is comparable

to those of previous works, and the actual values at the extremes of the measurement range are consistent with expectations. Counting errors on the wire screens ranged from about 10 percent at radon concentrations above 20 pCi-liter^{-1} and low CN concentrations to a high of 122 percent at 7 pCi-liter^{-1} and high CN concentration. Large sample volumes were necessary to minimize counting errors, and these were obtained by maintaining high sample flow rates. There is no evidence of deviation from expected screen collection efficiencies at the high sample flow rates used. Estimates of f_a by the backup filter method were generally consistent with those made with the wire screen, but had higher uncertainties at low radon concentrations and high CN values and a larger resultant data spread. For this reason, the results from backup filter measurements were not used as evaluation criteria.

Measurement results in Chamber C were somewhat different than expected. Where the Chamber A results and previous work demonstrated a rapid, exponential-shaped drop in f_a with a rise in CN concentration, the unattached fraction values in Chamber C

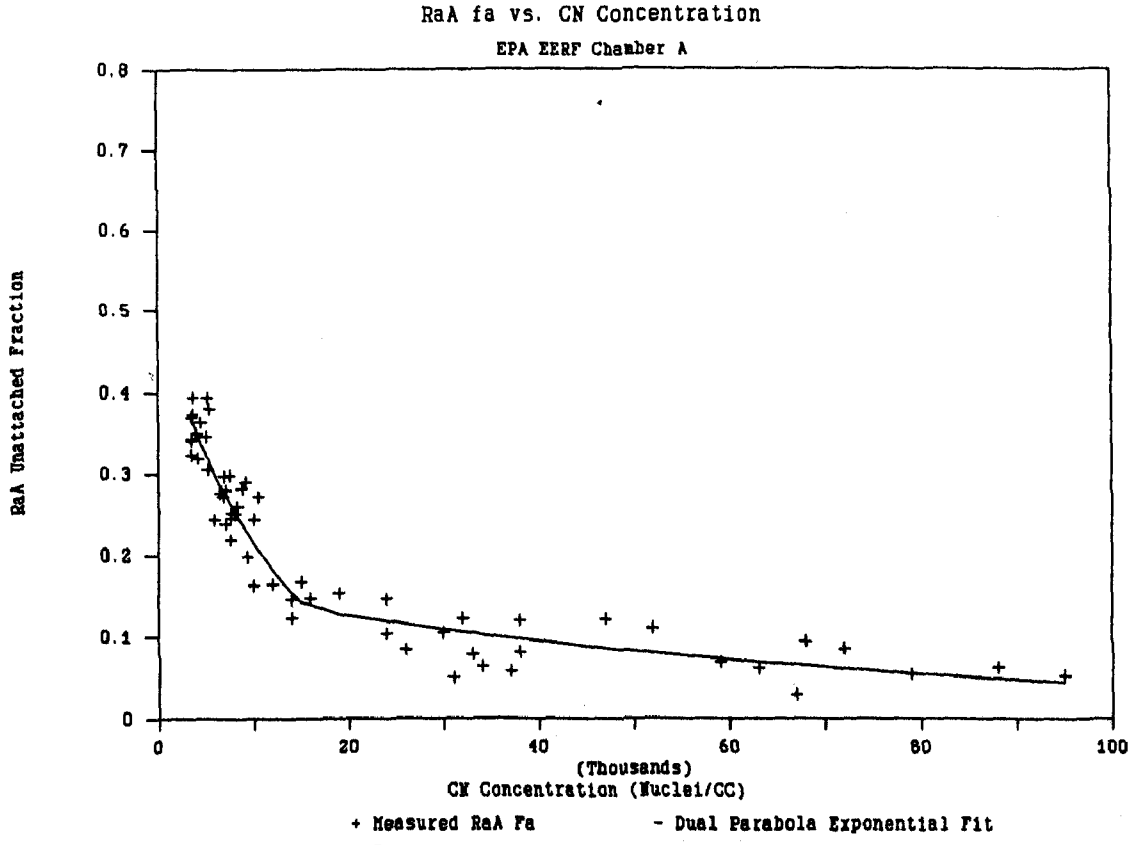
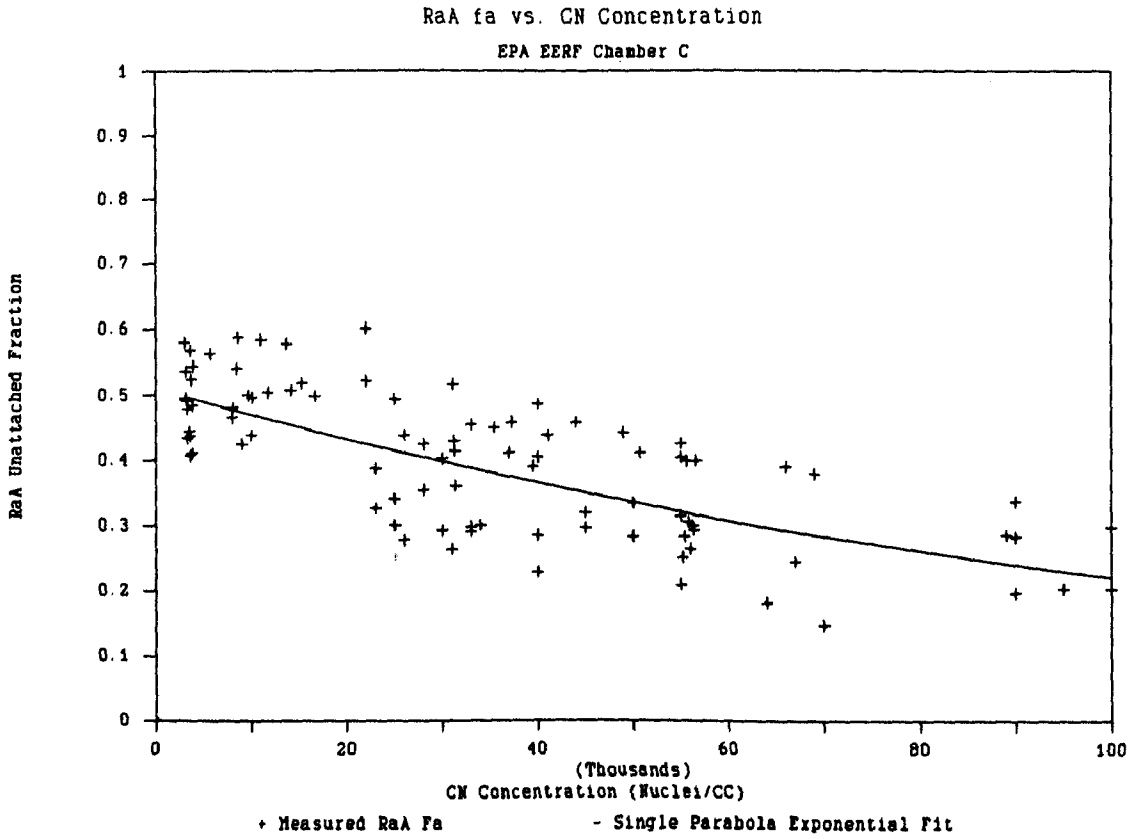


Figure 3. Chamber A f_a vs. CN Concentration

dropped at a much slower, nearly linear rate as CN values rose (see Figure 4). The actual f_a values at all condensation nuclei concentrations were somewhat greater than predicted by previous work or measured in Chamber A. As with measurements from Chamber A, backup filter results did not provide sufficiently consistent confirmation of f_a estimates to be used as evaluation criteria.

Several theories are currently under consideration regarding the unusual shape of the f_a versus CN curve in Chamber C. The most likely cause is related to source age and introduction method. In Chamber A, the radon gas is introduced through a recirculation plenum, where it mixes with the radon and CN throughout the chamber. In Chamber C, the radon gas is introduced though a single pass inlet plenum, where it is mixed with the CN. The gas/CN mixture is diffused across a large perforated particle board, and is then assumed to exhibit laminar flow through the chamber. The total time between introduction of gas and CN is about 30 minutes before sampling. While this time allows for sufficient ingrowth of RaA, it may not be sufficient to allow adequate combination of



Fugure 4. Chamber C f_a vs. CN Concentration

RaA and CN for representative f_a versus CN analysis. Further evidence of the limited mixing time is demonstrated by the low fraction of RaB and RaC'measured in both the screen and reference filter analyses.

In addition to differences in measured f_a which may result from the introduction method, it is probable that the measurements suffer from a bias in sample particle size. The unattached radon decay products in Chamber C are likely to be much smaller than those in Chamber A due to the age of the source at the time of sampling. It is also possible that the different chemical properties of the condensation nuclei affect the rate of combination with the RaA.

The data obtained from measurements in Chamber C is valuable for the purposes of determining counting errors and f_a measurement reproducibility, but not for confirmation of screen collection efficiencies. Screen counting errors ranged from 4 percent at high radon/low CN conditions to 38 percent at low radon/high CN conditions.

CONCLUSIONS

Several specific conclusions can be drawn from the analysis results, while several additional questions now require addressing. It is clear that simple wire screens can be used successfully as collection devices for free RaA within a reasonably well known set of limitations. Sampling with wire screens and backup filters does not provide more reliable results in difficult measurement circumstances. Sample accuracy at low radon/high CN concentrations is improved by increasing the sample volume. Screen collection and counting efficiencies appear to be consistent with that predicted by previous works.

Sampling with the wire screens at very low radon concentrations or very high condensation nuclei concentrations is likely to yield large counting uncertainties. To improve counting statistics under those conditions, large sample volumes are required, necessitating high sample flow rates or long sample durations. The same effect can be achieved with larger screens and high sample flow rates, as long the counting equipment can accomodate the larger screens. Due to the short half-life of RaA, sample durations longer than 10 minutes do not appreciably improve counting uncertainties.

In an effort to better understand the differences in results from EERF's Chamber C and expected results, measurement work will

continue there, focused on proving (or disproving) the previously stated hypothesis.

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Session III-P1:

Panel Session on QA/QC of Measurement

QUALITY ASSURANCE AND QUALITY CONTROL IN THE RADON MEASUREMENT INDUSTRY

WHAT IS MISSING?

by: Terry E. Howell
Radon Reduction & Testing, Inc.
Atlanta, Georgia 30329

ABSTRACT

Since the release of survey results by the consumer advocacy group BUYERS-UP, QA/QC has been the buzz word among everyone involved with Radon/Radon Decay Product measurement. This paper will present a discussion in generality of the roles of Quality Assurance and Quality Control in the Radon Measurement Industry, the strengths and weaknesses of QA/QC in radon measurement and the significance of research in QA/QC for radon measurement.

What are the important issues of quality assurance and quality control in radon measurement and...

WHAT IS MISSING?

INTRODUCTION

> " The EPA has established the National Radon Measurement Proficiency (RMP) Program. This quality assurance program enables participants to demonstrate their proficiency at measuring radon and radon decay product concentrations."

Everyone involved in radon measurement has read this statement - it appears in almost every EPA publication dealing with radon measurement and radon measurement protocols. But do we understand its meaning? Do we truly understand the purpose of the RMP Program and it's significance to the Radon Measurement Industry? If so, then why has the RMP Program left us stumbling blindly in a state of confusion, distrust, and uncertainty that is near chaos? Why do we hear statements from measurement companies that..."The RMP is really a certification program"... or ..."The EPA wants to control the Radon Industry"... or ..."The EPA has to do more than test our accuracy once every twelve to eighteen months"...? And why are consumer organizations and state agencies saying ..."the EPA must make the RMP Program tougher and more viable"...? WHAT IS MISSING?

DOLLARS AND SENSE

Perhaps the most important thing missing in Quality Assurance and Quality Control in the Radon Measurement Industry is simply that we do not understand the real reason for a QA/QC Plan. The reason that successful businesses develop and maintain Quality Assurance and Quality Control programs is because the results directly influence the bottom line of their Profit and Loss Statements. And it makes good business **SENSE** to pay close attention to those factors that directly influence the **DOLLARS.**

Statistical studies indicate that most businesses fail during their first three years and that the overwhelming majority of those failures are the result of one thing-Under Capitalization. The part that is really disturbing to financial analysts, lenders and investors is not that new ventures fail because of under capitalization but that most of the failures did not realize that they were under capitalized. They had never developed a Business Plan. If they had the cash flow projections in the plan would have identified the level of capitalization necessary to sustain them until they were profitable.

Developing and implementing a quality assurance plan and a quality control plan is nothing more than one integral part of our business plan. The results directly influence the two most important areas of our business plan--**The Financial Statement** (Reserves for Returned Goods and Services), and **The Marketing Plan** (in competitive, educated markets Quality Assurance may be the only advantage and therefore the core of the entire marketing strategy).

So how do we develop a Quality Assurance and Quality Control Program? The first step is to understand what the terms mean.

> **Quality Assurance** is the warranty or guarantee of quality and durability for goods and services that we impart to our customers.
>
> **Quality Control** is the system of inspections, testing and other measures that we employ to insure that claims against our warranty do not exceed anticipated levels (Reserves for Returned Goods and Services) and thereby become detrimental to our business financially.

As we can see these terms mean two entirely different things. But while they are not synonymous one is meaningless without the other.

Once the commitment has been made, the development of a Quality Assurance/Quality Control Plan is very much like the development of other portions of the business plan. We must perform a series of exercises and projections that when combined with an honest and objective evaluation of our capabilities and resources will provide us with all the elements of the plan.

ELEMENTS OF QUALITY ASSURANCE

The following are some of the important areas that should be considered in developing the QA Plan. More areas may need to be addressed depending on the complexity of the business structure or the scope of the project to be addressed.

1. Identify the benefits of a warranty. Will it give a competitive edge or is it mandatory for the market or the project? Will there be an implied warranty in the absence of a written warranty.

2. Identify the risks of a warranty.

3. Identify the acceptable level of exposure. To what degree will or extent will the warranty cover. Are there specific requirements?

4. Identify how failures will be handled. What will be the procedures for handling claims? What resources and reserves will be necessary to handle the predicted number of claims?

5. Formulate the written warranty. **YOUR ASSURANCE OF QUALITY.**

6. Designate personnel responsible for implementation of the program. **The Quality Assurance Officer**

7. Formulate a plan for insuring that claims on the warranty do not exceed the acceptable limits and become detrimental to the business. **The Quality Control Plan.**

8. Plan for catastrophic events. General, Product, Professional Liability Insurance. Is it available? Is it cost effective?

9. Seek out opportunities to demonstrate your capabilities or the effectiveness of your product. Are their existing programs available through consumer or government agencies that compare products or services to established or acceptable standards and publish their findings?

10. Monitor your performance. Evaluate performance in intercomparison programs. Evaluate number of warranty claims. Evaluate the impact on the market position.

ELEMENTS OF QUALITY CONTROL

The QC Plan will generally be technical in nature but must be designed to conform to the needs of the QA Plan. It is after all the primary insurance policy against warranty claims. The complexity of the plan will vary in proportion to the complexity of the work being performed. The following are some of the important areas to address.

1. Identify the task. This involves a thorough understanding of the over all scope of the project. The project should be divided into as many sub-categories as possible and then the portion or portions of the work to be performed should be identified.

2. Identify all the variables or factors within the task that will affect the performance of the work. What effect will each have? Can adverse effects be neutralized, standardized, or must they be accepted as is? How will each be handled?

3. Identify how the task will be performed. What equipment, methods or procedures will be utilized? What are the underlying theories or principals of operation of each? What factors or variables will affect the performance? How will these variables be handled? What are the maintenance and/or calibration requirements of the equipment? Is the method or equipment appropriate for the task? Is it practical?

4. Identify personnel requirements. What are the background requirements? What specific training will be required? What will their responsibilities be?

5. Identify the control options for each variable. What are the advantages and disadvantages of each? Are they appropriate? Are they practical? Are they effective?

6. Identify acceptable limits for precision and accuracy? Are there industry standards? Do they meet the needs of the Warranty?

7. Establish procedures for performance checks. How will each be performed? How often? How will failures be handled?

8. Adopt the plan. Put it in writing. Make it official by adoption at directors meeting.

9. Implement the plan. Make performance checks. Document the results.

10. Review performance. Were schedules met? Were performance specifications met? Were they sufficient to meet the needs of the warranty.

THE SIGNIFICANCE OF RESEARCH

Lets take a look at the process for developing a QC Plan.

Where do we get the information to identify the strengths and weaknesses of our detectors, methods, and instrumentation?

How do we discover the variables that influence our detectors, methods, and instrumentation?

How do we evaluate the effects of these variables?

In short, **where do we acquire all the information** that enables us to

1. Select the appropriate detector, method and instrumentation to perform the measurement.

2. Identify control options for the variables that influence performance.

3. Establish acceptable limits for precision and accuracy.

4. Establish procedures for performance checks.

Research is the very core of a Quality Control Program. If we do not understand the underlying principles of operation, capabilities and limitations, and the cause and effect of variables on our detectors, methods, and instrumentation we will never be able to develop much less implement an effective Quality Control Program.

WHAT IS MISSING?

What is missing? **An honest objective evaluation of ourselves as an industry.** We are unqualified, and untrained. We are unaware that there is chaos in our industry due to lack of confidence in ourselves and the absence of public trust. We are out of control, completely unable to offer an industry wide assurance of quality. Financial analysts have labeled us as a low tech, low potential industry and therefore unworthy of investor confidence. Anyone with a credit card and a business card can be a radon measurement expert before the sun goes down!

What is missing? **Responsibility.** We have lost sight of the scope of the project and its implications. We are dealing with an environmental public health issue. We have a moral obligation to be correct. Our clients are making decisions concerning the lives of themselves, their families, and in the case of public buildings, the general population, solely on the information are giving them.

What is missing? **Training.** No more awareness and overview training. We must have method specific and task specific training that incorporates an understanding of the underlying principles and theories of operation with method and task specific QA/QC procedures. Trained, qualified personnel is essential for quality control because it reduces the largest potential for error--**human error due to ignorance.**

What is missing? **Proper evaluation of our methods, detectors, and instrumentation.** An evaluation of our methods, detectors, and instrumentation in accordance to specific criteria will identify their capabilities and limitations, the measurement purpose for which they are most appropriate, the influence of variables, and the sources of potential error. This would lead to the establishment of specific QC procedures for each. Combined with a seal of approval, this would enable us to have the highest possible confidence in our measurements, and would work to restore public trust by acting as a deterrent to companies taking detectors and instrumentation to the market place without industry evaluation and acceptance.

What is missing? **Accreditation.** An industry wide quality control check to insure that our personnel have received the proper method specific training and demonstrated their understanding of the radon measurement principals and quality control procedures they will be utilizing. An accredited industry establishes an assurance of quality for all and is necessary if we are to squelch the "Tin-Man" image we presently portray.

What is missing? **Verification.** An inspection of our QC procedures, performance, and documentation to insure that the industry is practicing the daily, weekly, and monthly quality control necessary to provide a maximum assurance of quality deserving of the public trust.

What is missing? **Definition.** The EPA must define what it feels its role in radon QA/QC is. Is it political? Is it regulatory? Is it industry support? Is it consumer protection? Is it competition? The EPA must define what it expects from the radon industry in straight forward "tell it like it is language" not ambiguous statements filled with political rhetoric. More importantly, the radon industry must define its role. We must decide just what we expect from the EPA. Do we want government regulation of our business?

Both the radon industry and the EPA need to take a hard look at the RMP Program together and define it for what it is. Is it really a quality assurance program? If so where is the guarantee and to whom is it being conveyed? Is it a certification program that state and local agencies should use as minimum requirements (as many already do)? If so, what or whom is being certified? Perhaps it is time that the EPA stopped calling the RMP a quality assurance program. More importantly, **is it time the radon industry stopped calling it a quality assurance program.**

What is missing?

AN INDUSTRY WIDE ASSURANCE OF QUALITY AND THE CONTROLS TO BACK IT UP!

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

LOT-TO-LOT VARIATION IN THE PERFORMANCE OF PASSIVE ALPHA TRACK RADON DETECTORS

by: David L. Wilson
Oak Ridge National Laboratory
Oak Ridge, Tennessee

WITHDRAWN BY AUTHOR

COMPARISON OF THE ELECTRET-PASSIVE RADON MONITOR SYSTEM WITH CHARCOAL CANISTERS IN CONTROLLED ENVIRONMENTS

by: James R. Summers, William D. Nicholas, and
Robert L. Clemons
Analysas Corporation
Environmental Technology Division
101A East Tennessee Avenue
Oak Ridge, Tennessee 37830
and
David L. Wilson
Martin Marietta Energy Systems, Inc.
Health and Safety Research Division
Oak Ridge National Laboratory
Post Office Box 2008
Oak Ridge, Tennessee 37831-2008

ABSTRACT

The Electret-Passive Environmental Radon Monitor (E-PERM) was exposed to calibrated chamber radon concentrations of background, ~ 5, and ~ 60 pCi/l. The results were used to derive a statistical model to define the variability of measurements taken with the E-PERM. These results were compared with the manufacturer's calibration curves. The E-PERM and charcoal canisters were exposed simultaneously in a known chamber radon concentration held at a constant level. Results from the quality check of the manufacturer's calibration of the E-PERM validated the accuracy of the reported values. Data from the E-PERM and Femto-TECH are in agreement, while the data obtained with the charcoal canisters has a greater variability at identical radon concentrations.

INTRODUCTION

With recent interest in indoor radon concentrations, inexpensive radon screening methods have been developed. Charcoal canisters were the first to be developed and are currently the most widely used short-term (2 - 7 days) screening method. Because the charcoal canister relied on existing technology and the demand for an inexpensive and simple screening method was great, charcoal canisters quickly became an accepted radon screening method. However, studies have shown that charcoal canisters are not an ideal screening method (1-3). Sensitivity to environmental parameters such as temperature and humidity and the radon absorption characteristics of charcoal have lead to uncertainties in the results from screening with charcoal canisters. Even with this information known, screening with charcoal canisters was still widely conducted because alternative short-term screening methods were not available. Efforts were made to modify the design of charcoal canisters and improve performance. However, the large number of manufacturers of charcoal canisters and the wide variety of canister designs and materials used further confounded the validity of the results.

Technological advances with electret, a piece of dielectric material exhibiting a quasi-permanent electrical charge, have resulted in the development of a new short-term radon screening method (4-8). The device is called the electret passive environmental ^{222}Rn monitor (E-PERM), and it is currently manufactured and marketed by Rad Elect Inc.[1] The E-PERM device has shown little sensitivity to the environmental parameters that affect charcoal radon absorption. However, only a limited amount of test data are available from the literature on the performance of E-PERM and comparison of the two screening methods. Thus, radon screening with E-PERM is not yet widely practiced (compared with charcoal canisters).

The purpose of this study is to conduct an evaluation of the E-PERM's radon screening capabilities and a comparison of radon screening results between E-PERM and charcoal canisters in controlled radon environments. Recommendations for further studies also result from this research.

BACKGROUND

It is important to note the differences in how both methods accomplish the task of measuring indoor radon concentrations. The following discussion provides a brief description of each method and summarizes the advantages and disadvantages of each.

[1]Rad Elect Inc., 5330 J Spectrum Drive, 270 Technology Park, Frederick, Maryland 21701, (301) 694-0011.

CHARCOAL CANISTERS

The charcoal canister method takes advantage of the characteristic of activated charcoal to strongly adsorb radon. Activated charcoal also desorbs radon. An equilibrium is reached between the adsorbing and desorbing of radon and, thus, indoor radon is actively determined. As a result of this behavior, the measured radon concentration from the charcoal canister has much the same character as a "grab sample", which gives an instantaneous concentration biased to the radon level at the end of the test period. A study by UNC Geotech reported discrepancies of up to 40 percent from charcoal canister measurements due to variations in radon concentrations (1). Because indoor radon concentrations have been found to have significant diurnal variations, this could lead to large discrepancies between measured radon and actual radon concentrations for the exposure period.

Studies have found that radon absorption is sensitive to environmental parameters such as temperature, humidity, and air movement (1,3). The ability of charcoal to absorb radon increases as temperature decreases, and at very low temperatures (-78°C) charcoal becomes almost an infinite sink for radon. Discrepancy between measured radon and actual radon concentrations could occur as a result of the temperature dependence. Activated charcoal has a strong affinity for water vapor; as humidity increases, the ability for radon absorption decreases, which can result in a false negative. Air movements have a tendency to enhance the absorption, which can result in a false positive. Bocanegra and Hopke also found that air-borne organic contaminants can lower the dynamic adsorption coefficient of activated charcoal (3).

Despite these characteristics, the charcoal canister method is capable of measuring radon concentrations within EPA's acceptable precision (±25%), especially under ideal conditions. This precision under ideal conditions is further illustrated by the large number of radon testing companies that have passed Round 6 of the EPA Radon Proficiency Program. These tests are carried out in controlled environments, holding radon concentrations at a relatively constant level. Charcoal canisters used in this experiment were obtained from an EPA-listed radon tester. For the purposes of this research, the manufacturer will remain anonymous.

E-PERM

The E-PERM device is a small 40-ml interval container made of a semi-conducting material with a charged electret attached to the bottom of the container. The electret is made of Teflon® FEP[1] and is initially charged to 750 volts. The device is "turned on" by a spring loaded plunger that, when opened, allows the diffusion of radon through a filtered opening. When

[1]Teflon® FEP fluorocarbon manufactured by E. I. du Pont de Nemours and Company, Inc., Wilmington, Delaware 19898.

closed, radon gas can no longer diffuse into the internal volume of the container, and the plunger covers the electret. With the E-PERM turned off, the surface potential of the electret remains constant. The chamber behaves much like an ionization chamber. As the diffused radon gas undergoes decay, ions are generated. Negative ions (mostly electrons) are collected on the positively charged electret. The electret loses surface potential as a result of the deposition of ions. The loss in surface potential is proportional to the activity of the ionizing radiation (7,9). The device is also sensitive to background gamma radiation. Failing to account for background radiation could result in overestimating radon concentration by as much as 1 pCi/l.

Because the electret acts as an integrator of radon activity over time, the resulting measurement provides an average radon exposure during the test period. Integrated radon concentrations are a more useful diagnostic tool for radon testers because further actions are based on average exposures.

Kotrappa[14] found that E-PERM detectors showed little sensitivity to humidity (8). However, E-PERM detectors have shown a sensitivity to temperature. The surface potential of an electret will have a lower voltage when the electret is cold. The lower reading, because of the different expansion coefficients of the Teflon and electret holder material (10), will be interpreted as a result of radon activity and result in a false positive. The problem can be avoided if the reading of the electret (before and after the test) is performed within the temperature range of 65°F to 85°F and electrets are allowed at least 30 minutes to equilibrate to room temperature. Temperature variations during the testing period and travel will not affect the results.

Another potential problem that could misrepresent the measured radon concentrations is improper handling of the electret during the reading. Dust or dirt within the E-PERM chamber or from the air that contacts the electret will result in a lowering of the surface potential. Touching the electret will also lower the surface potential. The result would again be a false positive. The imposition of proper handling procedures in a clean environment will minimize the occurrence of false positives.

Even with special precautions in handling and reading of electrets, the E-PERM can give false positives. Out of the 80 E-PERM tests conducted in this study, one E-PERM gave a false positive result. EPA guidance states that false positives are preferred to false negatives (11). It is unlikely that the E-PERM system will give false negatives. False positives will be identified when confirmatory follow-up measurements are made with the E-PERM system.

Another benefit of the E-PERM system is that uncertainty due to varying manufacturer's specifications is limited because the E-PERM system is produced solely by Rad Elect, Inc. However, there are uncertainties associated with variations in the materials used to construct the E-PERM device. These variations are small and have not proven to contribute significantly to errors. The results from Analysas' participation in

Round 6 of the EPA Radon Measurement Proficiency Program had a standard error of 7%.

EXPERIMENTAL PROCEDURE

E-PERM detectors and charcoal canisters were exposed to radon in a 17-m^3 Hot Pack walk-in chamber located at the U.S. Department of Energy's Oak Ridge National Laboratory. Radon levels within the chamber are created from the National Bureau of Standards Traceable Radium Sources. A filtered air exchange, which is relatively radon free, is used to achieve a given radon concentration. To create background levels, the radium sources were not used, and only the filtered air supplied to the chamber. Radon concentrations within the chamber were determined by the use of four Femto-TECH Model RT310F Radon Trackers. The chamber was recently calibrated by the EPA Radiological Health Branch in Montgomery, Alabama. Resulting concentrations from the four Femto-TECHs were recorded by data loggers every 30 minutes. The results were then averaged over the time of the exposure to arrive at a time weight average that was used as the actual radon concentration of the chamber.

Three exposures were carried out in the chamber. The first exposure was with E-PERMs exposed at background levels. Subsequent runs exposed E-PERMs and charcoal canisters simultaneously. For each exposure, 20 of each device were exposed. Radon levels were allowed to stabilize before the radon detectors were placed. After exposure, the detectors were closed and returned to Analysas. The surface potential of the electrets was measured and radon concentrations calculated from manufacturer's calibration curves. Charcoal canisters were returned to the vendor for analysis following the vendor's specified handling protocols.

The results of the three E-PERM exposures were then used to verify the manufacturer's calibration curves. Results from a simultaneous exposure with E-PERMs and charcoal canisters were used for a comparison of the two radon measurement methods.

RESULTS

The following section provides the results of the study. The data are reported in two parts. The first gives the results from a check of the E-PERM manufacturer's calibration curve, and the second compares the E-PERM and charcoal canister methods.

QUALITY ASSURANCE STUDY OF MANUFACTURER'S CALIBRATION CURVE

The results of the E-PERM calibration study are shown in Figure 1. The solid line represents a linear regression through data from 60 detectors exposed at three radon concentrations. Interval estimates for this line

(α = 0.05) are also shown. The broken line is the calibration curve estimated by the manufacturer. Calibration factors were calculated by assuming the Femto-TECH data as actual values and calculate back to obtain the calibration factor.

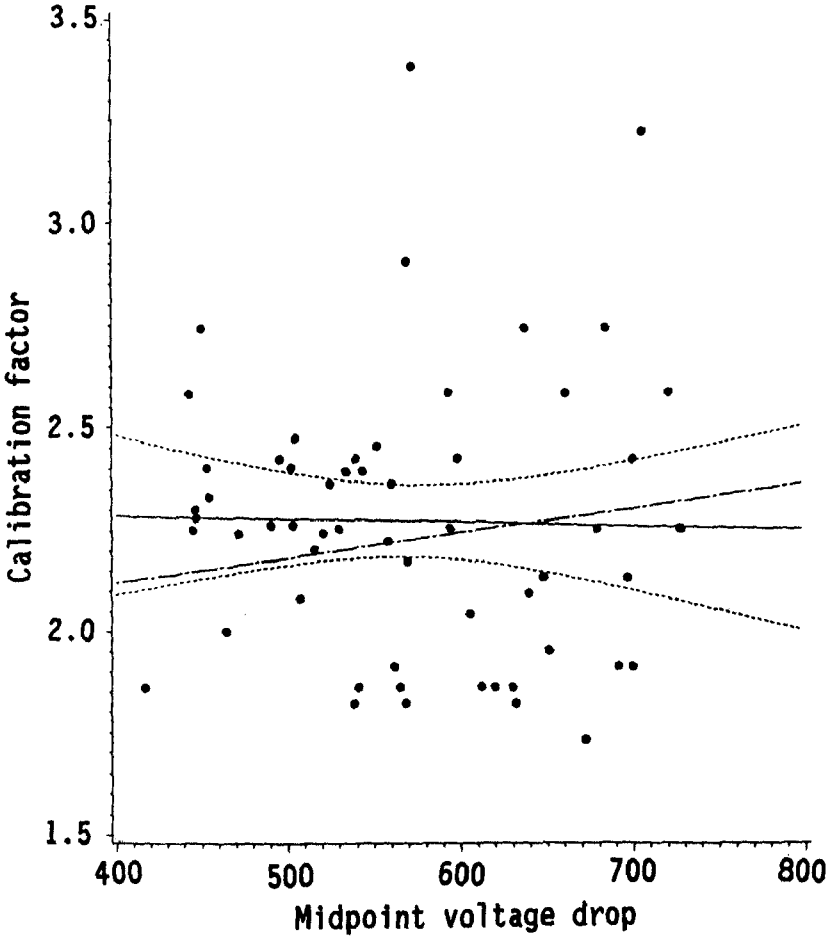

Figure 1. E-Perm calibration curve.

Solid line is linear regression through E-PERM data (n = 60). Dashed line is confidence interval (α = 0.05) about the regression line. Broken line is manufacturer's calibration curve.

Data used to generate Figure 1 were from E-PERM and Femto-TECH (shown on Table 1) and data from a background exposure. The results of the background exposure were 0.7 pCi/l for the Femto-TECH and 0.83 pCi/l with a standard deviation of 0.22 pCi/l for the E-PERM monitors. A simultaneous exposure of 300 alpha-track detectors was conducted at the background level, with a resulting average radon concentration of 0.68 pCi/l.

COMPARISON OF CHARCOAL CANISTER

Simultaneous exposures were conducted at radon concentrations of approximately 5 and 90 pCi/l. Table 1 summarizes the results from each run.

DISCUSSION

A correlation coefficient of 0.39 was calculated for the data plotted in Figure 1. This weak correlation coefficient shows that the E-PERM device responds linearly over the voltage range of the electrets used in the study. The weak correlation coefficient also means the slope of the calibration should be essentially zero. The slope of the manufacturer's calibration curve is 6×10^{-4}.

The quality assurance check of the calibration curve verifies the manufacturer's reported values. The manufacturer's curve falls within the 95% interval estimate from the estimated calibration curve. The data agree quite well even though a limited number of data points were generated from the study.

Using the Femto-TECH data as actual radon levels, both E-PERM and charcoal canister methods performed well for run 1. E-PERM detectors were more precise (accomplishing a standard deviation of 0.35 compared to 1.25 for charcoal canisters), while charcoal canisters were more accurate. Of the 20 charcoal canisters exposed at a radon level of 5.23 pCi/l, two canisters reported a false negative. Values of 1.5 pCi/l were reported for both false negatives.

For the second exposure, E-PERM and Femto-TECH data closely agreed with one another. However, charcoal canisters reported values that, on the average, were twice that of the assumed actual values. At this time the cause of the error is uncertain. The manufacturer was contacted, and various reasons for the discrepancy explored. No apparent cause for the error could be identified.

As a quality check, an E-PERM was exposed in radon levels of 132 pCi/l in the off position. After a two-day exposure, no loss in surface potential was measured.

The conclusion of the study was that the E-PERM device performed well, closely matching the precision of the Femto-TECH device. The quality of the charcoal canister's performance varied, with accurate readings at low radon level and grossly overestimated radon levels at high levels. The next phase

TABLE 1. RADON CONCENTRATIONS FOR SIMULTANEOUS EXPOSURES WITH CHARCOAL CANISTERS AND E-PERM DETECTORS

	E-PERM		Charcoal canister		Femto-TECH*	
	Rn concentration (pCi/l)	Standard deviation	Rn concentration (pCi/l)	Standard deviation	Rn concentration (pCi/l)	Standard deviation
Run 1	4.20	0.35	4.81	1.25	5.23	0.23
Run 2	66.65	3.12	133.12	7.25	63.62	3.01

*Used as the actual radon concentration.

of the research is to conduct simultaneous exposures of E-PERM detectors and charcoal canisters under varying radon concentrations. The varying radon concentrations will attempt to mimic the natural diurnal variation usually seen for indoor radon levels. Results from this study will be available in the coming months.

The work described in this paper was not funded by the U. S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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A RADON CHAMBER COMPARISON OF ALPHA TRACK DETECTORS OVER A RANGE OF EXPOSURES

by:

William M. Yeager, Keith A. Daum, and Robert M. Lucas
Center for Environmental Quality Assurance
Research Triangle Institute
Research Triangle Park, North Carolina 27709

Lisa Feldt
Office of Radiation Programs
U.S. Environmental Protection Agency
Washington, DC 20460

Edwin Sensintaffar and Sam Poppell
Eastern Environmental Radiation Facility
U.S. Environmental Protection Agency
Montgomery, Alabama 36109

Mike Clarkin
Camroden Associates
Oriskany, New York 13424

Three models of alpha track detectors (ATDs) were exposed in EPA radon chambers to obtain estimates of precision and bias for the National Residential Radon Survey (NRRS). Exposures in this study ranged from 365 to 7300 pCi/L-day, plus blanks. These exposures correspond to annual average concentrations of 1 to 20 pCi/L, the range expected in most U.S. residences.

Ten ATDs of each model were studied at 12 exposures. The mean and standard deviation of the reported values for each model were calculated and compared with the continuously monitored chamber concentrations to determine the bias and precision at each exposure. Results of this analysis were discussed with the vendors, who took corrective actions. Changes in track counting procedures and calibrations improved detector performance.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Alpha track detectors (ATDs) have been used for many years to measure the concentration of radon in soil gas, ambient air, and indoor air (1). ATDs are small (2 cm^2) sheets of plastic or film which are damaged by heavily ionizing charged particles (e.g., alpha particles from the decay of radon and its progeny). After exposure, the damaged material is preferentially etched by a caustic solution, leaving a scar in the plastic which is visible under an optical microscope. These scars or tracks may be counted by a trained person or by a computerized image analysis system. The density (number/area) of tracks is proportional to the exposure to radon. Exposure is the product of time and concentration. Previous studies have shown that track density is linear over a wide range of durations and concentrations (2,3,4). Given the duration of the exposure (days), the average concentration of radon (pCi/L) may be calculated. The sensitivity of an alpha track detector depends on its volume and geometry, as well as on the characteristics of the plastic and the etching process. Figure 1 illustrates an ATD. The sensitive plastic chip is enclosed in a cylindrical plastic housing, which has several openings in one end to allow radon to enter. A filter keeps out dust and radon progeny. Detectors are packaged in "radon-proof" foil bags to prevent exposure before placement.

The National Residential Radon Survey (NRRS) is measuring annual average radon concentrations in a statistical sample of about 5000 homes around the United States. The primary goal of the Survey is to estimate the distribution of radon concentrations in homes in the U.S., particularly the fractions of homes with annual average radon concentrations exceeding threshold levels of 4.0 and 10.0 pCi/L. The Survey is designed to estimate the fraction exceeding 10.0 pCi/L with an uncertainty, or relative standard error, of 50% or less if the fraction of homes is at least .005.

Radon concentrations will be measured using ATDs exposed for 12 months in the homes. This study of alpha track detectors was performed to evaluate which models could meet the precision and bias requirements of the NRRS. Information from this study was used in the design of the subsample of homes for the Survey which have co-located ATDs. The results of those 12-month measurements will be used to estimate the precision of the Survey measurements.

FACTORS AFFECTING THE QUALITY OF ATD MEASUREMENTS

ATDs are calibrated by exposure to known concentrations of radon. The response factor determined by these exposures relates the track density to the product of radon concentration with exposure duration. Track densities are determined by dividing the number of alpha tracks by the area in which they were counted. These counting statistics may be modeled using the Poisson probability distribution (5). The random counting error is the square root of the total count. The net track density is the difference

between the observed track density (determined by counting the number of tracks per unit area) and the average track density (or "background"), found on unexposed material immediately after it is received:

Net track density = track density - background correction

This background represents surface defects from the manufacturing process, not other radiation damage during the exposure period. Thus the background correction is part of the calibration of the detectors; it is independent of exposure, but may vary with the age of the plastic.

Assuming the net track density is a linear function of radon exposure, the exposures of ATDs may be calculated by multiplying the net track density by the response of the plastic material to known exposures:

$$\begin{aligned} \text{Exposure} &= (\text{response factor})(\text{net track density}). \\ &= R\ (\text{trkden} - B) \end{aligned}$$

where R = response
trkden = track density
B = background

Now, given the duration of the exposure (days), the average concentration of radon (pCi/L) can be calculated:

Average concentration = exposure/exposure duration

At low exposures, counting statistics limit the precision of ATDs. Only a small portion of the detector (10 to 50 mm^2) is routinely analyzed, but precision at these exposures may be improved by counting a larger area. Also, the accuracy of the background correction limits the bias of ATD measurements at low exposures.

At high exposures, errors in the response factor limit the precision if the counting error is truly random. Relative precision should improve as the exposure increases, asymptotically approaching the precision in the response factor. The accuracy of the response factor limits the bias of ATD measurements at high exposures.

DESIGN OF THE STUDY

Four sets of 140 commercially available models of ATDs were purchased through Camroden Associates, a subcontractor which does a considerable amount of radon measurement in NY and NJ. This blind purchase was arranged so that the vendors would not recognize that they were being tested. After delivery of the detectors, one vendor requested that their detectors be returned because of unspecified quality control problems. That model was therefore dropped from the study. Camroden shipped the rest of the detectors to EPA's Eastern Environmental Radiation Facility (EERF) in Montgomery, AL, for exposure to known concentrations of radon. There were two radon

chambers at the EERF. The larger ($42m^3$) walk-in chamber was operated at a nominal concentration of 36.5 pCi/L; the smaller ($3.6m^3$) chamber, at 365 pCi/L. The study lasted for 20 days in each chamber. The radon concentration in each chamber was continuously monitored to assure a reasonably constant level. Grab samples of chamber air were taken daily to calibrate the continuous monitors.

Each detector was identified by a manufacturer's serial number, which was permanently marked on the sensitive plastic chip, on the outside of the detector housing, and on the "radon-proof" foil bag in which each detector was packaged.

The detectors were randomly divided into 14 groups of 30. All 30 detectors in each group (10 of each model) received the same exposure. The nominal exposures ranged from 365 pCi/L-d, simulating an annual average of 1 pCi/L, to 7300 pCi/L-d, simulating an annual average of 20 pCi/L. Table 1 shows the chamber concentrations and exposure times. Note that the nominal chamber concentrations were chosen so that the detectors were always removed from the chamber at the same time of day.

There were two groups of blank detectors. Group 1 remained outside the radon chamber. Immediately before these detectors were returned to their vendors for analysis, their foil bags were opened and they were packaged for shipment to Camroden in the same fashion as the exposed detectors. This procedure checked for additional exposure in transit after the detectors were removed from the chambers. Group 3 detectors were left in their foil bags and placed in the large chamber. After 20 days, these detectors were removed from the chamber, the bags were opened, and the detectors were immediately packaged for shipment to Camroden. This procedure checked whether the foil bags completely prevented radon exposure before they were opened.

EXECUTION OF THE STUDY

At the EERF, the detectors were removed from their foil bags and placed in numbered wire trays. Care was taken that the serial numbers of the detectors in a tray were not sequential; again, we hoped to prevent the vendors from recognizing that there had been anything unusual in the placement.

Radon concentrations in each chamber were continuously monitored by α-particle counters. In addition, grab samples of chamber air were taken daily in 125-mL scintillation cells, which were held for 4 hours before counting. The nominal concentration in the small chamber was 365 pCi/L; the average monitored concentration was 366.5 pCi/L with a variation in the counting rate corresponding to $\pm$9.0 pCi/L. The nominal concentration in the large chamber was 36.5 pCi/L; the average monitored concentration was 36.1 pCi/L with a variation in the counting rate corresponding to $\pm$0.9 pCi/L. The total error (counting plus calibration) in the monitored radon concen-

tration for each chamber is estimated to be less than 5%. The monitored exposure for each group of detectors was calculated from the hourly averages during the exposure period of that group.

EERF personnel removed the detectors from the chambers according to the schedule in Table 1. The detectors were allowed 1 hour to equilibriate to the low radon concentration in room air before they were packaged for shipment to Camroden Associates. During this time, the serial numbers of the detectors were recorded on a data sheet. The 10 detectors of each model in each group were wrapped together in two layers of aluminum foil and marked with the group number.

Camroden personnel unwrapped the detectors and filled in dummy start and stop dates on the labels attached to each detector. These dates indicated an exposure of at least 2 weeks, the minimum period for which detectors might reasonably be exposed in homes suspected to have high radon concentrations. Detectors were then individually wrapped in foil and returned to their vendors for analysis. Exposure reports were received by Camroden and sent to RTI for evaluation.

RESULTS

Table 2 gives summary statistics of the reported exposures for the blank detectors. Group 1 detectors remained in their foil bags outside the radon chamber until immediately before shipment back to Camroden. Group 3 detectors remained in their bags inside the large chamber (36.5 pCi/L) for 20 days. They were removed from the chamber and then from their bags immediately before shipment back to Camroden. The vendors of detectors A and C did not report quantitative values <30 pCi/L-d. Thirty-five percent of the model A blanks exceeded this detection limit, while only 15% of model C blanks did so. All reported readings for model B blanks exceeded 30 pCi/L-d. It appears that the "radon-proof" bags of detector B allowed some exposure before receipt and especially while Group 3 was inside the radon chamber. The bags for detector B reduced the radon exposures for Group 3 by approximately 25% compared to the Group 4 detectors, which were in the same chamber for the same period.

Table 3 shows the nominal and monitored chamber exposures, summary statistics of the reported readings, and the recovery (reported/monitored) for four representative exposure groups. Data for all 12 exposure groups are included in Figures 2 - 6. Minimum and maximum values which differ by more than a factor of 2 may indicate an error by the laboratory which processed the detectors. Other than checking the serial numbers used to match actual with reported exposures, no attempt was made to resolve these outliers. The laboratories were not asked to confirm any reported values.

Figure 2 shows the relative standard deviation (RSD = standard deviation/mean) versus the monitored exposure for each group of detectors. In general, detector C had the best precision; detector B, the worst. Large increases in the RSD over a small range of exposures may be due to a transi-

tion from automatic to manual counting. As track density increases, so does the probability of tracks overlapping. This leads to undercounting, because automatic counting equipment has difficulty resolving overlapping tracks. The human eye, however, can resolve individual tracks at much higher densities. Therefore, detectors exposed above some transition point must be manually counted. Precision decreases after this transition for two reasons: 1) a smaller area is counted, and 2) people have difficulty reproducibly counting 50 or 100 things, especially under a microscope. The vendor of detector A advised us that their transition was at about 1500 pCi/L-d, where we see the RSD increased from 12 to 52%. We speculated that the transition to manual counting for detector B was about 3000 pCi/L-d, where the RSD increased from 7 to 17%, but later determined this was not the case. There is no apparent transition point for detector C in Figure 2.

Figure 3 shows the recovery vs. the monitored exposure for each model of ATD. An ideal detector would have a recovery of 1.0 with a small error bar. Detector A was closest to the monitored value, and therefore was selected for use in the NRRS, although it was biased high in the middle range of exposures. Detectors B and C were biased low; detector B had the greatest bias and the worst precision.

FOLLOW-UP

In May 1989, RTI and EPA personnel met with representatives of the vendors of models A, B, and C to discuss their performance in this study and what could be done to improve it. The detectors used in the study were identified, but their individual exposures were not.

In June 1989, we visited the facility where model A detectors were manufactured and analyzed. The vendor explained that they had two computerized image analysis systems: "wide-field" and "conventional." The first scanned a larger area than the latter, but could not resolve individual tracks at exposures above 1500 pCi/L-d. The detectors in our study had been counted by this system, with a transition to manual counting at 1500 pCi/L-d. This resulted in a large decrease in precision, observed between exposure groups 6 and 7. The vendor had recounted all of their detectors used in our study. Instead of making the transition to manual counting at 1500 pCi/L-d, they had changed to their conventional image analysis system, which could resolve tracks up to about 4500 pCi/L-d. At that point, they made the transition to manual counting. There was no significant difference in the reported exposure of the detectors which were recounted in the same way as originally counted, but the detectors which were first manually counted and then recounted with the conventional image analysis system averaged 45% less exposure on the recount. After receiving the recount data, RTI provided the vendor with the monitored exposures for the individual detectors used in the study.

Figure 4 compares the first and second values reported for detector A. The average precision for detectors exposed between 1500 and 4500 pCi/L-d improved from 26% to 10%. Overall, the average precision (RSD) improved from 20% to 15%, while the average bias changed from +8% to -9%.

In June 1989, we also visited the vendor of detector B. This vendor explained that they did not use an image analysis system; all detectors were manually counted. Thus, the deterioration in precision which we had observed at around 3000 pCi/L-d for model B detectors was not due to a change in counting procedure. We then hypothesized that it might be due to a difference between the two persons who counted the detectors, but it turned out that all of these detectors had been counted by the same individual. This vendor agreed to recount all of their detectors used in our study. After receiving the recount data, RTI provided the vendor with the monitored exposures for the individual detectors. In addition to recounting, the second set of reported values for detector B reflected changes in both the response factor and the background correction. The vendor also advised us that there had been two typographical errors in the first report that affected the reported values, and that two more detectors were damaged and should not have been reported.

Figure 5 compares the first and second values reported for detector B. Average bias improved from -35% to +6%; average precision (RSD) from 24% to 19%. The bias of detector B is now less than either of the other two detector models included in this study; however, the precision is still the poorest of the three, especially for low exposures ($<$ 1800 pCi/L-d or 5 pCi/L-yr).

After our site visit, vendor B began using an image analysis system to count their ATDs. Vendor B now uses this system to count all detectors. Because they spanned a wide range of exposures, detectors from our performance evaluation study were used to calibrate this system.

Table 4 shows the reported exposures of blank detectors determined from recounts. These may be compared with the values in Table 2. The reported exposures for model B detectors in group 3 are even higher than before. This vendor now reports that problems with sealing the foil bags in which the detectors are packaged have been solved.

Table 5 shows the reported exposures of non-blank detectors determined from recounts. These may be compared with the values in Table 3. The overall average recovery for all 12 groups of exposed detectors is 0.91 $\pm$ .14 for model A, 1.06 $\pm$ .19 for model B.

REGRESSION ANALYSIS

If the reported values of spiked detectors are precise, but biased, a linear regression of the monitored values on the reported values may be performed to check the calibration of the detectors. Table 6 shows the

results of such a regression (based on recounts for models A and B). A least squares fit was used to determine the coefficients S and I of the equation

$$\text{Monitored} = S*\text{Reported} + I$$

Ideally, monitored = reported or S = 1.0 and I = 0.0. A value of S significantly different from unity indicates that the response factor is off. It appears that the response factor is low for detector C and high for detectors A and B. This may be related to the fact that the vendors' calibration exposures are usually less than 500 pCi/L-d. Calibration exposures should span the anticipated range of field exposures. On the other hand, a value of I significantly different from zero indicates that the background correction is not accurate. It appears that the background correction is too large for model A and too small for model C, but about right for model B. This correction is based on the observed track density of freshly manufactured plastic or film, not on counts of completely assembled detectors which have been distributed throughout the country, sealed, and mailed back to the vendor. Note that at low exposures, the errors in the response factors and background corrections for detectors A and C tend to cancel out. Tables 7 and 8 summarize the interpretation of the regression coefficients in general and for the specific models in this study.

These regression coefficients may be used to correct the calibration coefficients:

$$\begin{aligned} \text{Monitored} &= S*\text{Reported} + I = S*R*(\text{trkden} - B) + I \\ &= R'*(\text{trkden} - B') \end{aligned}$$

$$\text{where } R' = S*R$$
$$B' = B - I/(S*R).$$

Alternatively, the regression coefficients may be used to adjust the reported values:

$$\text{Adjusted} = S*\text{Reported} + I.$$

Figure 6 shows the results of such an adjustment for model C detectors. The model C detector was designed for non-residential applications. Although it had the best precision of the three models in our study, it showed significant bias. These two characteristics made it an ideal candidate for adjusting the reported values based on the regression described above. Overall, the average recovery improved from 0.742 ± .068 to 1.009 ± .099; the mean absolute relative error (MARE), from 0.261 to 0.078. Results of such an adjustment for the other models, which had worse precision and possible non-calibration problems, were less impressive.

CONCLUSIONS

Performance of measurement methods should be evaluated over the range of exposures anticipated, before field placement. The importance of various measures of performance depends on the intended use of the measurement. The decision to mitigate an individual home will be based on a few measurements; each one is important. It is not necessary, however, to be very accurate at high or low exposures. One would like to maximize measurement quality at the exposure corresponding to the decision of whether or not to mitigate the structure. The EPA guideline for mitigation is an annual average radon concentration of 4 pCi/L, or a 12-month exposure of 1460 pCi/L-d.

On the other hand, in a survey to determine the frequency distribution of radon concentrations, it is important to know the precision and bias over the entire range of exposures, so that the distribution can be adjusted on a statistical basis. For the NRRS, performance will be monitored by the use of blank, collocated, and spiked detectors which will be mixed with field exposed detectors returned for analysis.

In either case, the performance of alpha track detectors on a well-designed set of known exposures may be the basis of discussions with the vendor about suspected problems with calibration, processing, or counting. In some cases, calibration coefficients may be adjusted on the basis of a regression analysis of spiked detectors.

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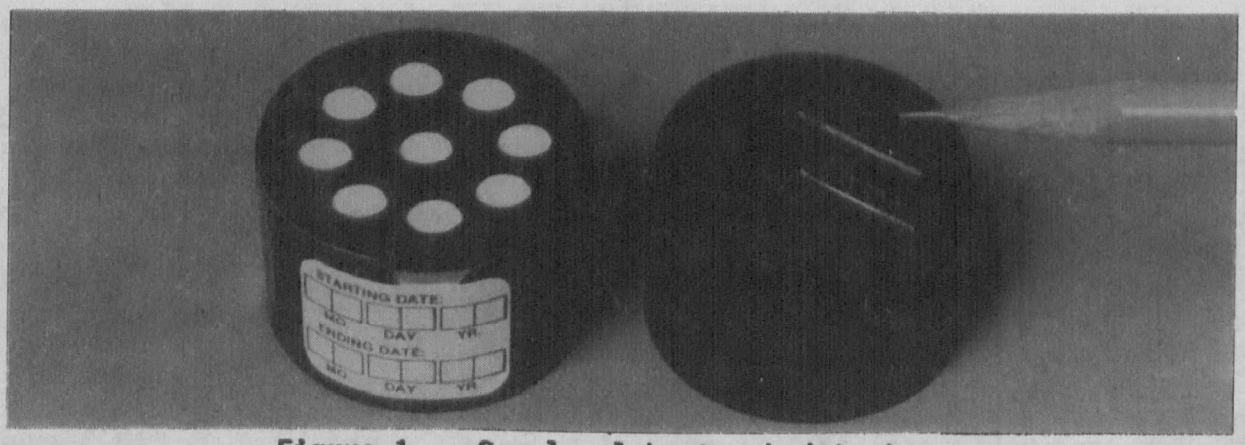


Figure 1. Sample alpha track detector.

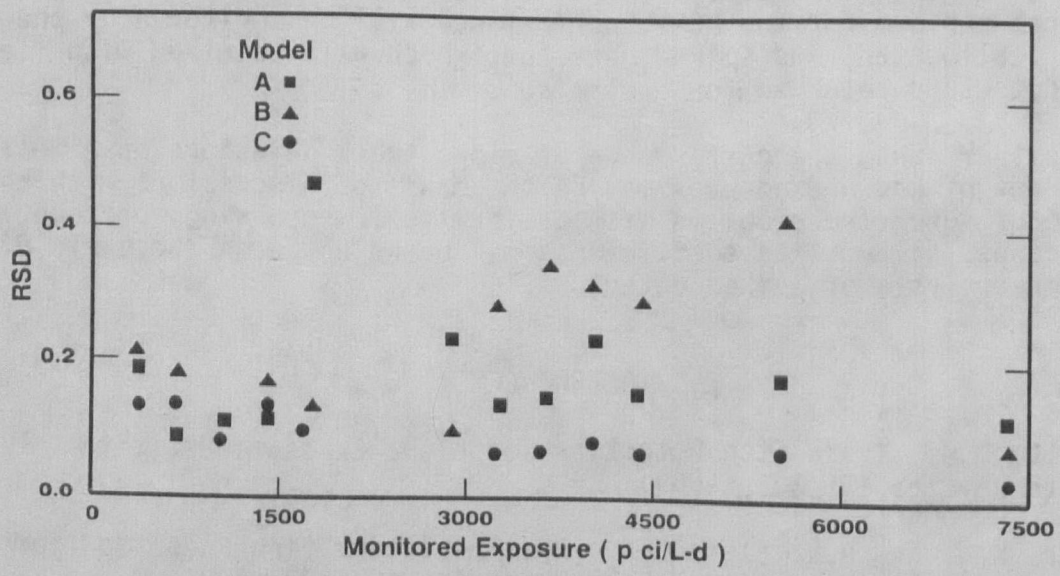


Figure 2. Reported precision of three ATD models.

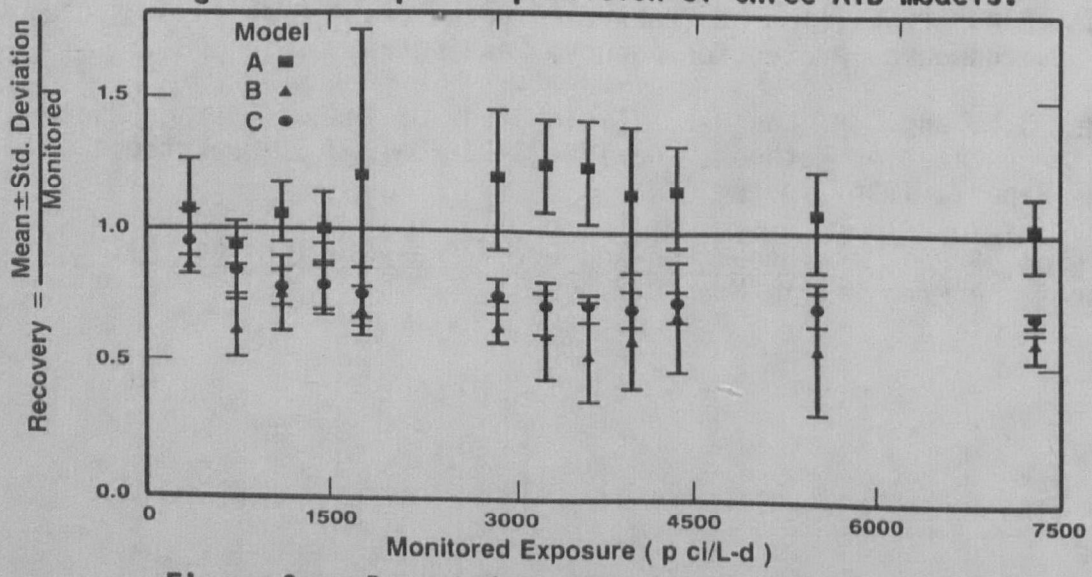


Figure 3. Reported vs. monitored radon exposures.

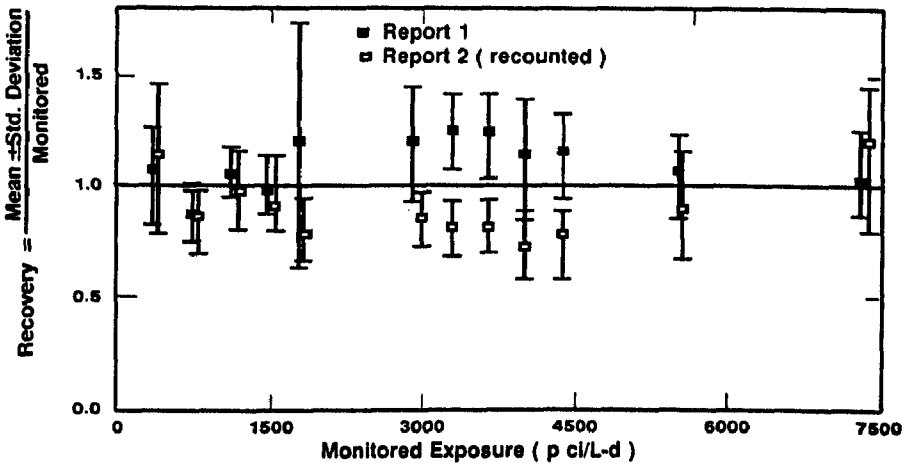


Figure 4. First and second values reported for Model A.

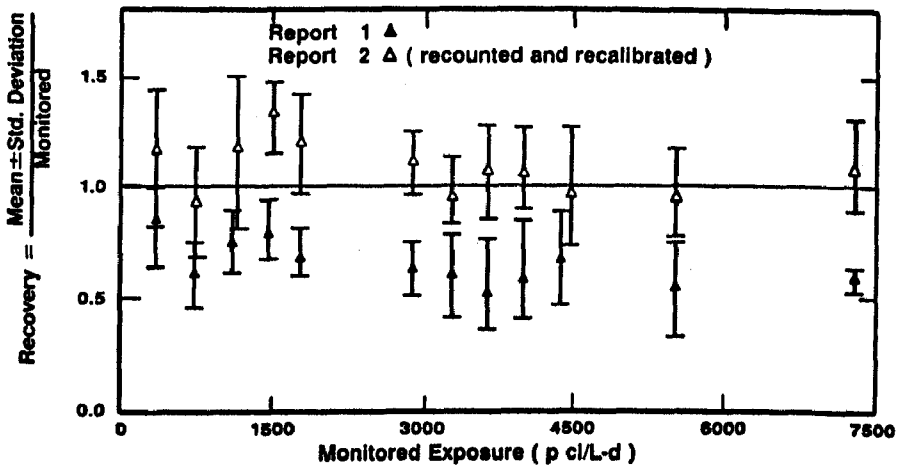


Figure 5. First and second values reported for Model B.

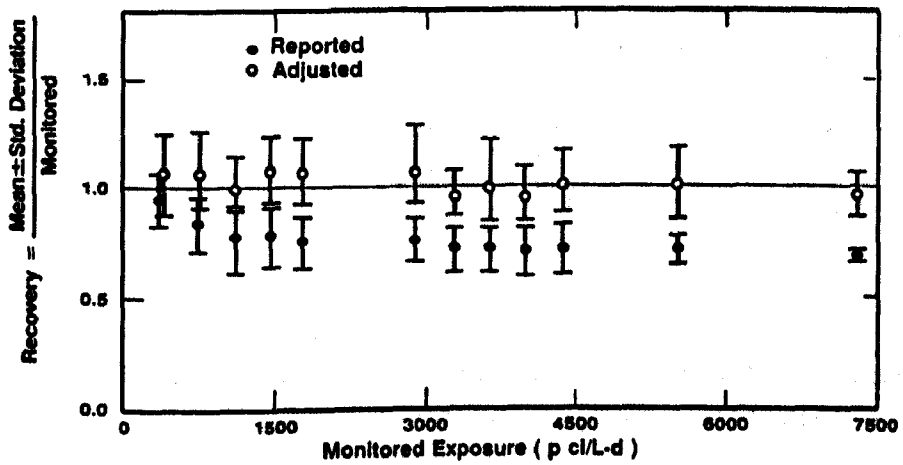


Figure 6. Reported and adjusted values for Model C.

TABLE 1. NOMINAL CHAMBER CONCENTRATIONS AND EXPOSURE TIMES

Group	Planned Exposure (pCi/L-yr)	Planned Exposure (pCi/L-d)	Chamber Conc. (pCi/L)	Chamber Time (days)
1	0	0	0 (unopened)	0
2	1	365	36.5	10
3	0	0	36.5 (unopened)	20
4	2	730	36.5	20
5	3	1095	365	3
6	4	1460	365	4
7	5	1825	365	5
8	8	2920	365	8
9	9	3285	365	9
10	10	3650	365	10
11	11	4015	365	11
12	12	4380	365	12
13	15	5475	365	15
14	20	7300	365	20

TABLE 2. REPORTED RADON EXPOSURES FOR BLANK DETECTORS

Group[a]	Model	Nominal	Reported[b] N>30	Reported[b] Median	Percentiles 90th	Percentiles 10th
1	A	0	4	24	56	9
	B	0	10	64	118	48
	C	0	2	<30	43	<30
8	A	0	3	15	52	5
	B	0	10	357	403	218
	C	0	1	<30	33	<30

[a] Group 1 detectors remained in foil bags outside the radon chambers. Group 8 detectors remained in foil bags inside a radon chamber for 722 pCi/L-d.

[b] Only values > 30 pCi/L-d were reported for model C.

TABLE 3. REPORTED RADON EXPOSURES

Group	Model	Monitored Exposure (pCi/L-yr)	Monitored Exposure (pCi/L-d)	Reported Exposure (pCi/L-d) Median	Mean	RSD	90th	10th	Recovery[a]
2	A	1.0	364	362	378	0.178	480	300	1.03 ± 0.16
	B			292	302	0.211	420	250	0.83 ± 0.17
	C			319	331	0.128	410	290	0.91 ± 0.12
6	A	3.9	1432	1372	1395	0.118	1590	1210	0.97 ± 0.12
	B			1147	1115	0.169	1320	750	0.78 ± 0.13
	C			1097	1088	0.124	1320	890	0.76 ± 0.09
10	A	10.0	3646	4494	4370	0.151	5150	2990	1.20 ± 0.18
	B			2020	1940	0.342	2820	510	0.53 ± 0.18
	C			2566	2569	0.076	2840	2260	0.70 ± 0.05
14	A	20.1	7330	7666	7387	0.124	8370	5570	1.01 ± 0.13
	B			4460	4407	0.086	4890	3850	0.60 ± 0.05
	C			5098	502	0.036	5230	4670	0.69 ± 0.02

[a] Recovery = (mean reported ± std.dev.)/monitored

TABLE 4. RECOUNTED RADON EXPOSURES FOR BLANK DETECTORS

Group[a]	Model	Nominal	Reported[b] N>30	Reported[b] Median	Percentiles 90th	Percentiles 10th
1	A	0	3	23	74	1
	B	0	10	47	109	19
3	A	0	4	25	82	11
	B	0	10	420	677	232

[a] Group 1 detectors remained in foil bags outside the radon chambers. Group 3 detectors remained in foil bags inside a radon chamber for 722 pCi/L-d.

[b] Only values > 30 pCi/L-d were reported for model C.

TABLE 5. RECOUNTED RADON EXPOSURES

Group	Model	Monitored Exposure (pCi/L-yr)	Monitored Exposure (pCi/L-d)	Reported Exposure (pCi/L-d) Median	Mean	RSD	90th	10th	Recovery[a]
2	A	1.0	364	366	387	0.231	563	298	1.06 ± 0.25
	B			399	404	0.259	570	288	1.11 ± 0.29
6	A	3.9	1432	1212	1331	0.167	1681	1060	0.93 ± 0.16
	B			1953	1847	0.133	2130	1480	1.29 ± 0.17
10	A	10.0	3646	3386	3173	0.168	3469	2623	0.87 ± 0.09
	B			3816	3744	0.166	4900	2926	1.03 ± 0.17
14	A	20.1	7330	7199	8042	0.275	12270	6253	1.10 ± 0.30
	B			7758	7672	0.159	9520	5770	1.05 ± 0.17

[a] Recovery = (mean reported ± std.dev.)/monitored

TABLE 6. REGRESSION COEFFICIENTS OF REPORTED VERSUS MONITORED EXPOSURES (MONITORED = S*REPORTED + I)

Model	S	I	R-square
A2	0.851 ± .033	690.7 ± 115.2	0.8505
B2	0.916 ± .026	149.8 ± 99.1	0.9154
C	1.456 ± .017	-107.1 ± 42.9	0.9845

TABLE 7. GENERAL INTERPRETATION OF REGRESSION COEFFICIENTS MONITORED = S*REPORTED + I

	R	I
small	S > 1.0	I < 0.0
large	S < 1.0	I > 0.0

TABLE 8. SPECIFIC INTERPRETATION OF REGRESSION COEFFICIENTS IN TABLE 6 MONITORED = S*REPORTED + I

Model	S --> R		I --> B		Comments
A2	< 1.0	large	> 0.0	large	cancel at low exposures
B2	< 1.0	large	> 0.0	satisfactory	
C	> 1.0	small	< 0.0	small	cancel at low exposures

A STUDY OF BATCH CALIBRATIONS ON 4" OPEN FACED CHARCOAL ADSORBERS FROM FOUR DIFFERENT MANUFACTURERS AND HOW THEY COMPARE TO EERF'S PUBLISHED CALIBRATION CURVES

by: Dallas L. Jones and T. E. Howell
Radon Reduction & Testing, Inc.
1587 Northeast Expressway
Suite 175
Atlanta, Georgia 30329

ABSTRACT

The EERF Standard Operating Procedures for Rn-222 Measurements Using Charcoal Canisters is quite explicit in stating that the calibration tables and curves published in this manual are only "typical calibration data that were developed for a particular batch of canisters from a supplier," and that "new calibration data are developed and used with each different lot of canisters."

In this study, batch calibrations were performed in our radon chambers on so called "EPA style" canisters from four different companies. We then prepared calibration tables and curves for each batch and compared them to the EERF published tables. It was our premise that EPA never intended for their curves to be considered "standard" and that with more and more companies manufacturing canisters, the calibration factors could certainly vary and might significantly differ from those derived by EERF several years ago.

INTRODUCTION

During the past year of promoting our calibration chamber services, it became apparent to us that many labs analyzing 4" open-faced cans are blindly using the calibration tables and curves programed into the software supplied with their instrumentation, without ever developing calibration curves of their own. In most cases, these curves were taken directly from the EERF publication. We were concerned that companies may be placing a false sense of security in calibration curves that may not apply to their particular canisters.

PRELIMINARY QUALITY ASSURANCE CHECKS

Prior to beginning our study, we chose to perform quality assurance checks on our calibration chambers by performing a current chamber mapping and an intercalibration with the EERF chamber in Montgomery. Radon Reduction & Testing chambers were designed to recirculate the radon-laden air to eliminate stratification of heavier radon gas to the bottom. We must be conscious however, that air flows are not excessive enough to cause over response by the charcoal adsorbers. (2)

CHAMBER MAPPING

To doubly insure that the radon concentrations within our calibration chambers is being evenly distributed and that air currents are not adversely affecting absorption, charcoal cartridges were placed on every shelf in each coded "bin" location, approximately 10 inches apart. The average radon concentration in Chamber 1 for this 24 hour mapping exposure was 33.3 pCi/l; the average radon concentration in Chamber 2 during the mapping exposure was 5.8 pCi/l.

Comparing calibration factors calculated for each detector exposed Chamber 1, we determined a standard deviation of less than 4%. Comparing calibration factors calculated for each detector exposed in Chamber 2, we determined a standard deviation of less than 8%. These differences are well within the standard deviations between charcoal detectors themselves and indicated to us that the chamber environment was consistent throughout. The slightly higher deviation between detectors in Chamber 2 were attributable to the low radon concentration in that chamber during mapping exposures. (Precision between canisters exposed at 6 pci/l would generally be lower than precision between canisters exposed at 33 pCi/l.) Another mapping in chamber 2 upon project completion at a higher radon concentration proved this to be the case. At 30 pCi/l the standard deviation between cans was slightly greater than 3%.

INTERCALIBRATION

On August 15, 1989 two Eberline RGM-3 continuous gas monitors used as back-ups for continuous chamber monitoring and for performing daily grab sampling to cross check the full time chamber monitors, were taken to Montgomery and intercalibrated with the EERF radon chambers. Comparing these monitors with our full time chamber monitors indicated agreement within 3% and no need for a calibration adjustment.

BATCH SELECTIONS

The charcoal canister batches used in this study were graciously supplied by four different laboratories or suppliers. Each batch exposed and analyzed will heretofore be referred to as Company Red, Company Green, Company Yellow and Company Pink; the laboratories suppling the canisters will not be identified by name. All canisters were 4" diameter open faced, described as EPA style canisters and supposedly assembled to EPA specifications described in the EERF publication. (1)

Upon receiving the cans, we randomly selected 5 from each batch of 100 to be analyzed for any differences and to determine the initial carbon weight and moisture content for an average can from each batch. Although all of the canisters were described by the laboratories providing them as EPA style canisters, analysis revealed certain obvious differences. Initial moisture contents of the carbon in each batch varied, even though all were within the EPA specification of less than 4%. The initial carbon weight of cans from Company Pink averaged 76.1 grams. (Remember EPA specs of 70 grams plus or minus 1 gram.) The average initial carbon weight and moisture content of the charcoal in each batch was determined to be the following:

Co.	RED	69.75g initial carbon wt.	2.8% initial moisture content
Co.	GREEN	69.54g initial carbon wt.	1.8% initial moisture content
Co.	PINK	76.10g initial carbon wt.	2.4% initial moisture content
Co.	YELLOW	70.66g initial carbon wt.	1.9% initial moisture content.

Of the 5 cans analyzed for each batch, initial moisture content was more inconsistent in those provided by Company Green and Company Yellow. Company Red's RECORDED initial weights were consistently 0.3 to 0.4 grams lighter than they were at the time we weighed them. It was assumed that the batch had picked up the extra moisture (through the tape) while being stored prior to our receiving them. Conversations with several canister manufacturers confirmed our belief that they can indeed gain several tenths of grams on the shelf if the storage environment is humid enough. One company reported he had cans that had gained nearly 0.7 grams while being stored for nearly a year.

Examination of the carbon in these cans revealed differences as well. Company Red and Company Green appeared to be Calgon carbon. Company Yellow

was much less dusty than typical Calgon carbon, but was otherwise the same in appearance. Company Pink carbon was different in density and appearance and was apparently not Calgon. (An anonymous telephone conversation with an unidentified representative from Company Pink did confirm that it was not Calgon carbon, but he assured me that the "adjustment factor curves are the same as published in the EERF Standard Operation Procedures Manual... the same as our Calgon Cans.")

PREPARATION & EXPOSURE SEQUENCE

After selecting 5 cans from each batch of 100 for determining initial carbon weight and moisture content, and selecting a background can from each batch, 90 cans from each batch were weighed and initial weights (to 0.1 gram accuracy) and serial numbers were logged on chamber data sheets. Chamber 1 was brought to 80% RH, Chamber 2 to 50% RH. Chamber 2 was adjusted to 20% RH the following week for the low humidity run. Radon concentration for the 6 days averaged 47 pCi/l for the 80% run, 29 pCi/l for the 50% run, and 25 pCi/l for the 20% run. During exposure periods, continuous chamber monitors were checked against daily grab radon samples. Chamber temperatures were maintained between 72 and 74 degrees F.

When radon concentrations and humidity in the chamber were stabilized at the desired conditions, a set of 30 canisters from each company were put into the chamber and exposed in the following manner; a group of five each for 1,2,3,4,5, and 6 days, respectively. The canisters were opened inside the chamber and the exact start times were recorded. This sequence was repeated at each of the three humidities, to complete our calibration run.

REMOVAL AND COUNTING

At the end of each 1 day period specified above, a group of 5 canisters were closed & taped inside the chamber and exact time was recorded. At least 3 hours were waited prior to counting to allow progeny to equilibrate with the radon.

After removal from the chambers, each canister was weighed again, recorded, and subsequently counted on our counting system. Background canisters from each batch and our "standard" can were counted prior to each counting period to determine our detector efficiency.

CALCULATION OF CALIBRATION FACTORS

A calibration factor was calculated for each canister using the following equation published in the EERF Standard Operating Procedures for Radon-222 Measurement Using Charcoal Canisters:

$$CF = \frac{\text{net cpm}}{(Ts)(E)(Rn)(DF)}$$

CF = Calibration factor, radon absorption rate (1/m), net
Net CPM = Gross counts per minute for the canister minus the background CPM for the detector for that day,
Ts = Exposure time of the canister in the chamber (minutes),
Rn = The Radon concentration in the chamber for the exposure period (pCi/l),
DF = Decay factor from the midpoint of exposure to the time of counting, which is calculated from

$$DF = e^{-\frac{.693t}{T\,1/2\ Rn(min)}}$$

t = time in minutes from midpoint of exposure to the start of counting and T 1/2 Rn(min) is the half-life of radon 222 (3.82 days) in minutes. (1)

GENERATION OF CALIBRATION CURVES

Calibration factors were derived from the equation above to generate two tables and two curves for each batch of canisters studied. One table relates calibration factors to weight gain (water) for a two day exposure (the optimum exposure time recommended by EPA for the EPA style 4" open face can). Data in these tables were used to generate the two day weight gain vs. calibration factor curves. NOTE: The weight gain curves generated for this study utilize only the known data points from the 2-day calibration exposures and were not extrapolated beyond known weight gains for two day exposures at 80% RH.

The second table relates exposure time to adjustment factors for the 20%, 50% and 80% humidity. The adjustment factors are used to modify the calibration factors for exposure times different from the desired 2 day exposure time. If a can is exposed for exactly 2 days, the adjustment factor is 1. (1)

RESULTS

The first and most obvious discovery made upon completing the calibration tables for all four batches was the Company Pink curves plotted considerably below the EERF curves and those for the other three companies. Calibration factors ran as much as 30% lower than EERF's at 20% RH. In fact, Company Pink's 20% RH curve runs similar to EERF's published 50% RH curve. This is not surprising when we consider that Company Pink's carbon was definitely not Calgon, although the manufacturer of this can told us the calibration curves would be " the same as his Calgon cans..., the same as the EERF curve." The different brand of charcoal made a tremendous difference in the calibration. The carbon used is not as sensitive as Calgon and is much more dense.

Remember it took approximately 76 grams to fill up the can. The difference in the carbon does not make it a bad can, however. The precision among detectors was as good as the other three companies and our initial inspection of the cans found them to be quite consistent in initial weight and moisture content. With the correct calibration tables these cans perform admirably.

Curves for the other three companies ran surprisingly close to the EERF curves through three days. This we found somewhat surprising because of the variance in the initial moisture contents.

Deviations from EERF published curves were greatest beyond 72 hours for Companies Red, Green, and Yellow. Calibration factors ran LOWER than EERF published curves at 20% RH and HIGHER than EERF's published curves at 80% RH. In other words beyond three day exposures the cans were somewhat less efficient than EERF's at 20% RH and somewhat more efficient than EERF's at 80% RH. These differences were enough to cause accuracy problems beyond 3 days if your software is utilizing the EERF tables. Using EERF data, these companies would be reporting much too high at 4-6 days at 80% RH exposures and much to low at 4-6 day 20% RH exposures.

A conversation with a representative of Company Yellow demonstrates this well. Upon re-analyzing the canisters we exposed from her company, her system utilizing EERF published curves computed a radon concentration for a can exposed for 6 days at 80% RH to be 88 pCi/l. The actual exposure concentration was 47.7 pCi/l. This is an error of over 80%. Upon changing her adjustment factor for 6 days at 80% from .018 to her new AF of .027 she read 58 pCi/l; much closer to the know concentration than before, but still not 47.7 pCi/l.

The remaining difference can be explained when one closely examines the EERF published 2 day weight gain curve. Our Company Yellow technician had not adjusted this curve to her new data and that certainly would account for some of her error. But the real problem lies in the extrapolated slope of the two day curve beyond the known weight gains at 80% RH for 2 days.

Assuming you had a batch of cans that matched the EERF data perfectly, you would know that a can exposed for 6 days at 80% RH and gaining 18 grams, would have a CF of .018. This is a known CF determined in the calibration run and is printed in the table. (1) But using EERF's tables and curves and the formula for computing a final calibration factor from them, you would end up with a final CF of .011, NOT .018. This is because the initial CF pulled from the 2 day weight gain table is .049. In order to compute to correct concentration, the initial CF would have to be .078. As the EERF weight gain curve slopes below .078 to .049 at 18 grams, the final CF drifts farther and farther apart from the known CF for those cans exposed for 3-6 days at 80% RH in the calibration run.

Simple logic would lead us to believe that herein lies the answer. If the initial CF never drifted below .078, we ewould always end up with a final CF equivalent to our known calibration factors. If we gained more weight than what our calibration cans did at 3-6 days at 80% RH, we would know that they

were exposed to a much greater humidity than 80% and we would simply throw them out.

But while this solves one problem, it creates another by not allowing for adjustment of CFs for those cans that perhaps gained only 10 grams in 6 days. Such a can was exposed to something less than 80% RH and would have a CF of somewhat higher than a can exposed for 6 days gaining 18 grams. If the weight gain curve becomes a straight line after the maximum 2 day weight gain at 80% RH, the can gaining 10 grams in 6 days would have the same CF as the can gaining 18 grams in 6 days.

We can only assume that this dilemma is one of the reasons EPA set their optimum exposure time at 2 days for the 4" open face cans.

CONCLUSIONS

Even though three of the four batches of EPA style canisters studied followed the EERF published curves reasonably well through 3 days, the curves were all somewhat different. Beyond 3 day, they were extremely different. We certainly believe our study supports EERF's recommendation that calibration curves be generated by every laboratory for THEIR detectors and that these be checked regularly with exposures to known concentration.

The dilemma regarding the 2-day weight gain vs. CF curve vs. troublesome and extremely important when analyzing cans with exposure times greater than 2 to 3 days. The curve in its present from "misses the mark" on KNOWN exposures at 80% RH, even with the EERF's own data, for exposures beyond 3 days. Changing it to a constant beyond the maximum weight gain for 2 day calibration exposures at 80% RH, would eliminate the error for 3 to 6 day known exposures at 80% RH, but would no adjust for the can that gained only 10 grams at 6 days rather than the 16 to 18 grams we know it would have gained at 80% RH. Perhaps the simplest solution would be to discard any measurement made for beyond 3 days and stay as close to the optimum 2 day exposure as possible.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

REFERENCES

1. Gray, D.J. and Windham, S.T., EERF Standard Operating Procedures for Radon-222 Measurement Using Charcoal Canisters, EPA 520/5-87-005, U.S. Environmental Protection Agency, Montgomery, Alabama, 1987, 29 pp.

2. Gray, D.J. and Windham, S.T., The Overesponse of Open Faced Charcoal adsorbers Used For Measurements Of Indoor Radon Concentrations, U.S. Environmental Protection Agency, Montgomery, Alabama.

TABLE 1. EXPOSURE TIME VERSUS ADJUSTMENT FACTORS FOR LOW, MEDIUM, AND HIGH HUMIDITY
COMPANY PINK VERSUS EERF PUBLISHED

EXPTIME	CO PINK LOW RH	EERF PUB LOW RH	CO PINK MED RH	EERF PUB MED RH	CO PINK HIGH RH	EERF PUB HIGH RH
24	0.126	0.137	0.117	0.132	0.105	0.116
24	0.122	0.143	0.108	0.137	0.105	0.125
24	0.130	0.141	0.109	0.132	0.105	0.118
24	0.126	0.135	0.106	0.126	0.105	0.117
24	0.121	0.138	0.121	0.127	0.101	0.118
48	0.092	0.107	0.082	0.096	0.065	0.077
48	0.095	0.110	0.080	0.102	0.066	0.082
48	0.090	0.105	0.081	0.097	0.067	0.076
48	0.093	0.101	0.082	0.094	0.067	0.076
48	0.091	0.105	0.084	0.098	0.067	0.078
72	0.075	0.087	0.063	0.075	0.048	0.048
72	0.072	0.091	0.061	0.079	0.048	0.051
72	0.070	0.088	0.063	0.075	0.046	0.051
72	0.070	0.083	0.062	0.073	0.049	0.046
72	0.072	0.085	0.061	0.075	0.047	0.049
96	0.057	0.074	0.053	0.058	0.035	0.035
96	0.057	0.080	0.051	0.062	0.036	0.034
96	0.058	0.075	0.051	0.059	0.036	0.033
96	0.060	0.074	0.054	0.057	0.036	0.033
96	0.060	0.075	0.054	0.060	0.037	0.034
120	0.050	0.070	0.045	0.051	0.030	0.023
120	0.052	0.073	0.045	0.054	0.028	0.025
120	0.052	0.071	0.043	0.051	0.029	0.024
120	0.052	0.069	0.045	0.050	0.030	0.023
120	0.053	0.071	0.045	0.052	0.029	0.023
144	0.049	0.064	0.039	0.045	0.022	0.018
144	0.048	0.068	0.039	0.047	0.022	0.019
144	0.049	0.064	0.038	0.047	0.022	0.016
144	0.051	0.062	0.038	0.044	0.018	0.018
144	0.048		0.039		0.025	

TABLE 2. WEIGHT GAIN VERSUS CALIBRATION FACTORS FOR A 2 DAY RUN

COMPANY PINK VERSUS EERF PUBLISHED

	COMPANY PINK		EERF PUBLISHED	
PERCENT HUMIDITY	WATER GAIN (g)	CF LITER/MIN	WATER GAIN (g)	CF LITER/MIN
20	-0.3	0.092	0	0.105
20	-0.3	0.095	0	0.101
20	-0.2	0.089	0	0.105
20	-0.2	0.093	0	1.110
20	-0.1	0.091	0	0.107
50	1.6	0.082	1.7	0.095
50	1.6	0.080	1.8	0.094
50	1.7	0.081	1.9	0.097
50	1.6	0.082	1.8	0.102
50	1.7	0.084	1.8	0.096
80	7.3	0.065	7.7	0.077
80	7.6	0.066	7.5	0.082
80	8.4	0.067	7.7	0.076
80	6.7	0.067	7.9	0.076
80	6.4	0.067	7.9	0.078

TABLE 3. EXPOSURE TIME VERSUS ADJUSTMENT FACTORS FOR LOW, MEDIUM, AND HIGH HUMIDITY
COMPANY GREEN VERSUS EERF PUBLISHED

EXPTIME	CO GREEN LOW RH	EERF PUB LOW RH	CO GREEN MED RH	EERF PUB MED RH	CO GREEN HIGH RH	EERF PUB HIGH RH
24	0.154	0.137	0.128	0.132	0.122	0.116
24	0.137	0.143	0.138	0.137	0.123	0.125
24	0.134	0.141	0.134	0.132	0.116	0.118
24	0.138	0.135	0.133	0.126	0.125	0.117
24	0.139	0.138	0.137	0.127	0.121	0.118
48	0.106	0.107	0.094	0.096	0.078	0.077
48	0.106	0.110	0.097	0.102	0.077	0.082
48	0.107	0.105	0.095	0.097	0.077	0.076
48	0.106	0.101	0.096	0.094	0.077	0.076
48	0.106	0.105	0.095	0.098	0.077	0.078
72	0.088	0.087	0.079	0.075	0.056	0.048
72	0.088	0.091	0.077	0.079	0.056	0.051
72	0.085	0.088	0.080	0.075	0.055	0.051
72	0.085	0.083	0.072	0.073	0.054	0.046
72	0.085	0.085	0.076	0.075	0.056	0.049
96	0.070	0.074	0.065	0.058	0.040	0.035
96	0.070	0.080	0.067	0.062	0.041	0.034
96	0.071	0.075	0.065	0.059	0.042	0.033
96	0.070	0.074	0.064	0.057	0.043	0.033
96	0.069	0.075	0.066	0.060	0.038	0.034
120	0.062	0.070	0.055	0.051	0.033	0.023
120	0.061	0.073	0.056	0.054	0.033	0.025
120	0.062	0.071	0.055	0.051	0.031	0.024
120	0.062	0.069	0.055	0.050	0.036	0.023
120	0.064	0.071	0.056	0.052	0.032	0.023
144	0.057	0.064	0.051	0.045	0.018	0.018
144	0.058	0.068	0.050	0.047	0.021	0.019
144	0.060	0.064	0.049	0.047	0.022	0.016
144	0.056	0.062	0.048	0.044	0.019	0.018
144	0.057		0.049		0.024	

TABLE 4. WEIGHT GAIN VERSUS CALIBRATION FACTORS FOR A 2 DAY RUN

COMPANY GREEN VERSUS EERF PUBLISHED

	COMPANY GREEN		EERF PUBLISHED	
PERCENT HUMIDITY	WATER GAIN (g)	CF LITER/MIN	WATER GAIN (g)	CF LITER/MIN
20	0	0.106	0	0.105
20	0.1	0.106	0	0.101
20	-0.3	0.107	0	0.105
20	-0.1	0.106	0	1.110
20	-0.1	0.106	0	0.107
50	1	0.094	1.7	0.095
50	1	0.097	1.8	0.094
50	1.4	0.095	1.9	0.097
50	1.4	0.096	1.8	0.102
50	1.4	0.095	1.8	0.096
80	7.3	0.078	7.7	0.077
80	8.3	0.077	7.5	0.082
80	8.4	0.077	7.7	0.076
80	9	0.077	7.9	0.076
80	7	0.077	7.9	0.076

TABLE 5. EXPOSURE TIME VERSUS ADJUSTMENT FACTORS FOR LOW, MEDIUM, AND HIGH HUMIDITY
COMPANY RED VERSUS EERF PUBLISHED

EXPTIME	CO RED LOW RH	EERF PUB LOW RH	CO RED MED RH	EERF PUB MED RH	CO RED HIGH RH	EERF PUB HIGH RH
24	0.155	0.137	0.139	0.132	0.117	0.116
24	0.155	0.143	0.141	0.137	0.116	0.125
24	0.158	0.141	0.143	0.132	0.119	0.118
24	0.154	0.135	0.139	0.126	0.122	0.117
24	0.156	0.138	0.145	0.127	0.116	0.118
48	0.110	0.107	0.100	0.096	0.071	0.077
48	0.110	0.110	0.099	0.102	0.071	0.082
48	0.111	0.105	0.102	0.097	0.068	0.076
48	0.112	0.101	0.100	0.094	0.069	0.076
48	0.111	0.105	0.099	0.098	0.071	0.078
72	0.087	0.087	0.077	0.075	0.052	0.048
72	0.086	0.091	0.078	0.079	0.048	0.051
72	0.087	0.088	0.077	0.075	0.049	0.051
72	0.088	0.083	0.079	0.073	0.054	0.046
72	0.087	0.085	0.080	0.075	0.050	0.049
96	0.070	0.074	0.063	0.058	0.035	0.035
96	0.068	0.080	0.065	0.062	0.035	0.034
96	0.068	0.075	0.064	0.059	0.035	0.033
96	0.068	0.074	0.065	0.057	0.040	0.033
96	0.072	0.075	0.064	0.060		0.034
120	0.066	0.070	0.054	0.051	0.028	0.023
120	0.064	0.073	0.054	0.054	0.028	0.025
120	0.064	0.071	0.056	0.051	0.029	0.024
120	0.065	0.069	0.054	0.050	0.031	0.023
120	0.068	0.071	0.054	0.052	0.028	0.023
144	0.055	0.064	0.046	0.045	0.021	0.018
144	0.056	0.068	0.047	0.047	0.022	0.019
144	0.055	0.064	0.048	0.047	0.022	0.016
144	0.056	0.062	0.046	0.044	0.021	0.018
144	0.054		0.046		0.022	

TABLE 6. WEIGHT GAIN VERSUS CALIBRATION FACTORS FOR A 2 DAY RUN

COMPANY RED VERSUS EERF PUBLISHED

	COMPANY RED		EERF PUBLISHED	
PERCENT HUMIDITY	WATER GAIN (g)	CF LITER/MIN	WATER GAIN (g)	CF LITER/MIN
20	-0.7	0.110	0	0.105
20	-0.5	0.110	0	0.101
20	-0.5	0.111	0	0.105
20	-0.5	0.111	0	1.110
20	-0.5	0.112	0	0.107
50	1.1	0.099	1.7	0.095
50	1	0.099	1.8	0.094
50	0.9	0.100	1.9	0.097
50	1.1	0.099	1.8	0.102
50	1	0.099	1.8	0.096
80	8.4	0.071	7.7	0.077
80	8.3	0.071	7.5	0.082
80	8.1	0.068	7.7	0.076
80	8.6	0.069	7.9	0.076
80	8.4	0.071	7.9	0.078

TABLE 7. EXPOSURE TIME VERSUS ADJUSTMENT FACTORS FOR LOW, MEDIUM, AND HIGH HUMIDITY
COMPANY YELLOW VERSUS EERF PUBLISHED

EXPTIME	CO YELLOW LOW RH	EERF PUB LOW RH	CO YELLOW MED RH	EERF PUB MED RH	CO YELLOW HIGH RH	EERF PUB HIGH RH
24	0.149	0.137	0.137	0.132	0.119	0.116
24	0.143	0.143	0.122	0.137	0.119	0.125
24	0.149	0.141	0.125	0.132	0.125	0.118
24	0.151	0.135	0.125	0.126	0.125	0.117
24	0.143	0.138	0.141	0.127	0.113	0.118
48	0.107	0.107	0.091	0.096	0.077	0.077
48	0.105	0.110	0.094	0.102	0.076	0.082
48	0.106	0.105	0.091	0.097	0.076	0.076
48	0.017	0.101	0.096	0.094	0.078	0.076
48	0.105	0.105	0.093	0.098	0.074	0.078
72	0.083	0.087	0.072	0.075	0.054	0.048
72	0.085	0.091	0.073	0.079	0.057	0.051
72	0.084	0.088	0.071	0.075	0.056	0.051
72	0.084	0.083	0.073	0.073	0.057	0.046
72	0.083	0.085	0.077	0.075	0.056	0.049
96	0.067	0.074	0.062	0.058	0.043	0.035
96	0.067	0.080	0.064	0.062	0.038	0.034
96	0.069	0.075	0.063	0.059	0.043	0.033
96	0.070	0.074	0.060	0.057	0.043	0.033
96	0.067	0.075	0.064	0.060	0.041	0.034
120	0.061	0.070	0.053	0.051	0.034	0.023
120	0.057	0.073	0.053	0.054	0.035	0.025
120	0.059	0.071	0.053	0.051	0.033	0.024
120	0.058	0.069	0.051	0.050	0.036	0.023
120	0.061	0.071	0.050	0.052	0.033	0.023
144	0.055	0.064	0.047	0.045	0.026	0.018
144	0.054	0.068	0.045	0.047	0.026	0.019
144	0.057	0.064	0.047	0.047	0.029	0.016
144	0.058	0.062	0.047	0.044	0.027	0.018
144	0.056		0.046		0.028	

TABLE 8. WEIGHT GAIN VERSUS CALIBRATION FACTORS FOR A 2 DAY RUN

COMPANY YELLOW VERSUS EERF PUBLISHED

	COMPANY YELLOW		EERF PUBLISHED	
PERCENT HUMIDITY	WATER GAIN (g)	CF LITER/MIN	WATER GAIN (g)	CF LITER/MIN
20	0	0.107	0	0.105
20	0.1	0.105	0	0.101
20	0	0.106	0	0.105
20	0.2	0.107	0	1.110
20	0.2	0.105	0	0.107
50	1.2	0.091	1.7	0.095
50	1.1	0.094	1.8	0.094
50	1.2	0.091	1.9	0.097
50	1	0.096	1.8	0.102
50	1.1	0.093	1.8	0.096
80	6.2	0.077	7.7	0.077
80	7.2	0.076	7.5	0.082
80	7	0.076	7.7	0.076
80	6	0.078	7.9	0.076
80	6.2	0.074	7.9	0.078

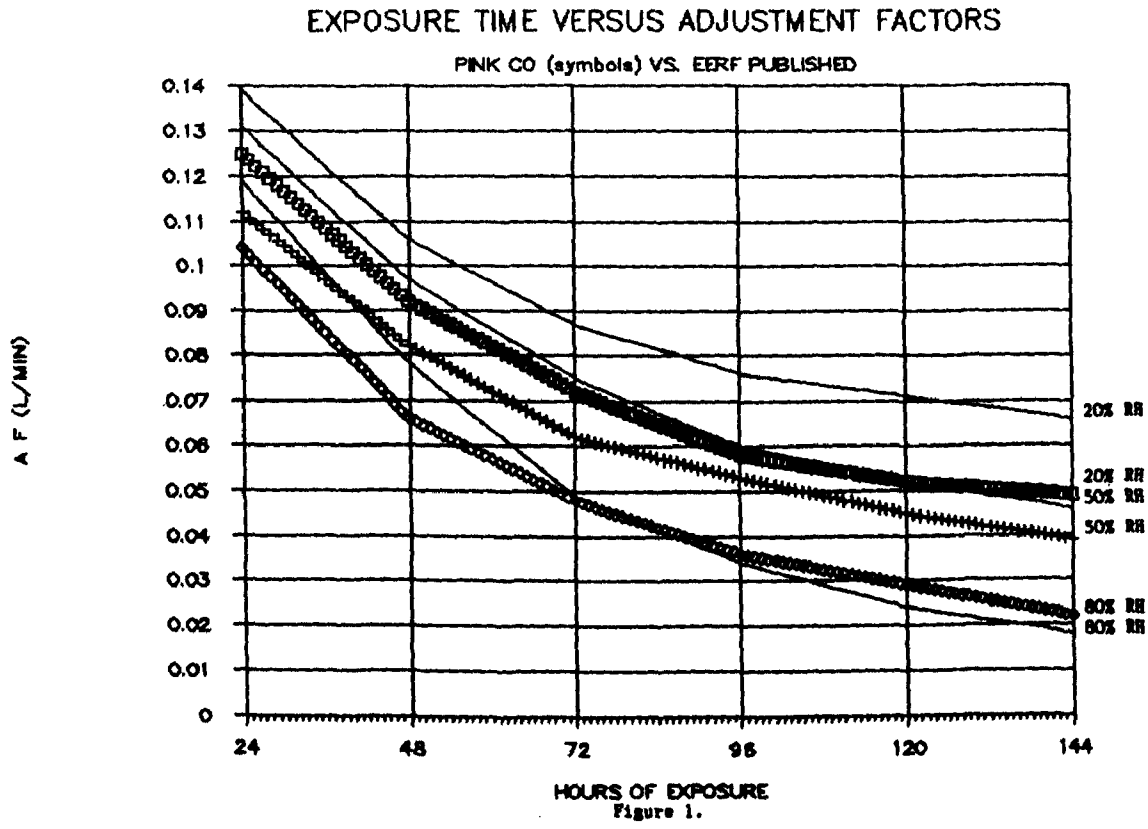
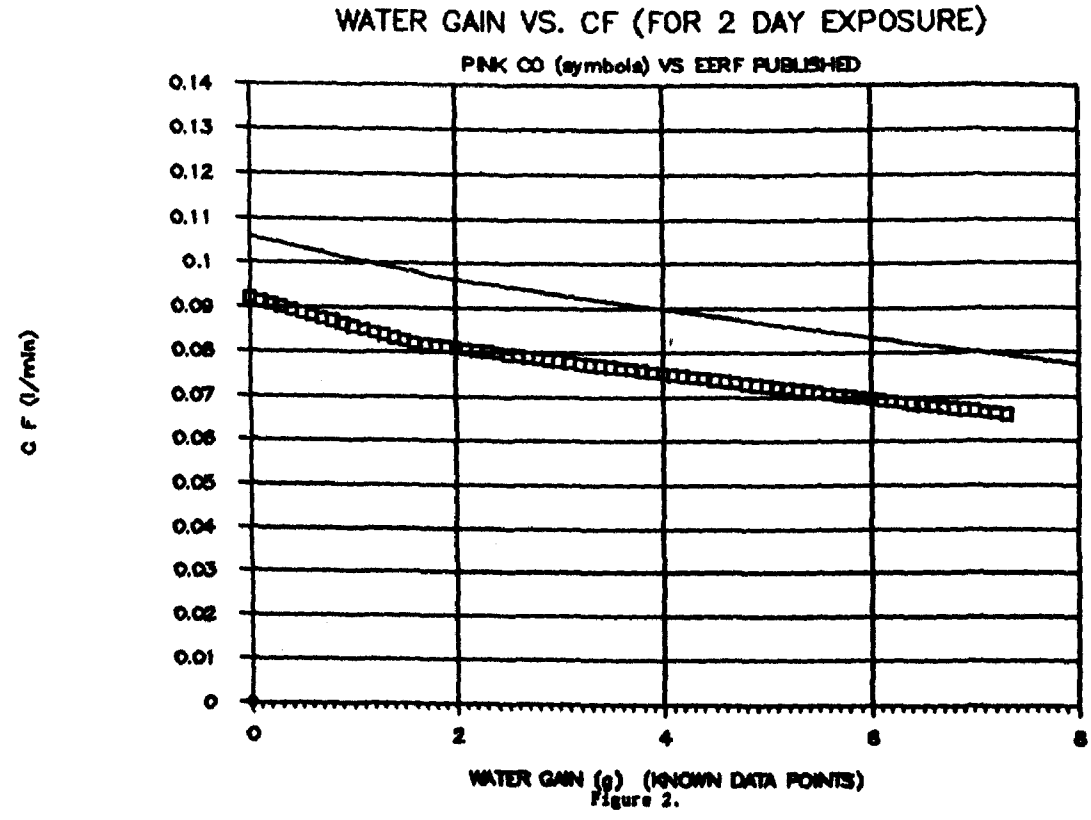


Figure 1.



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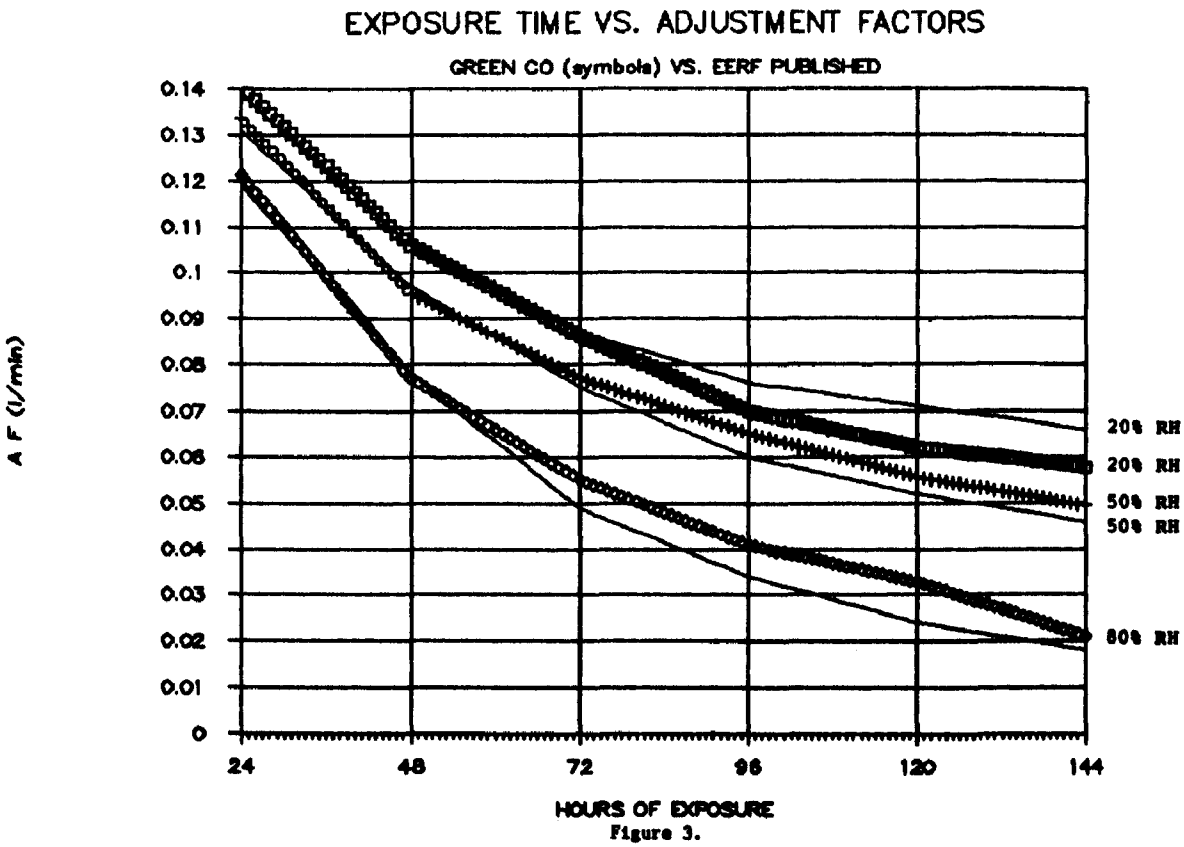

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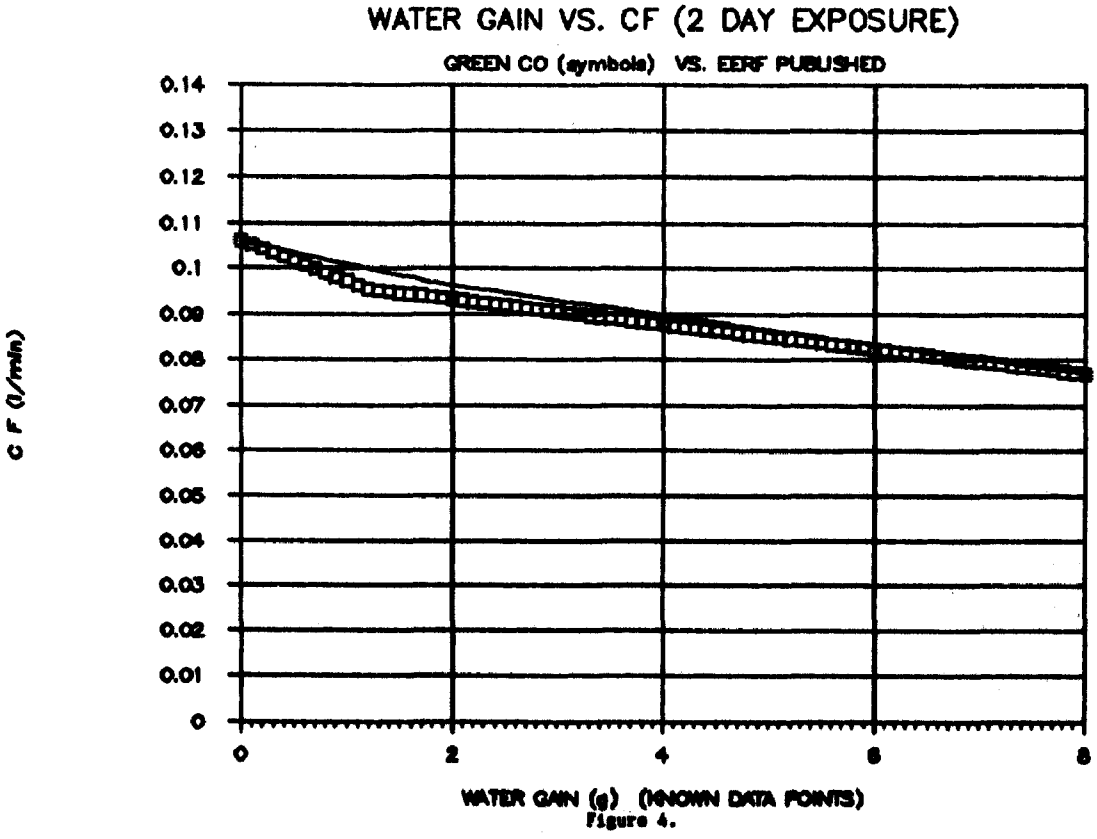

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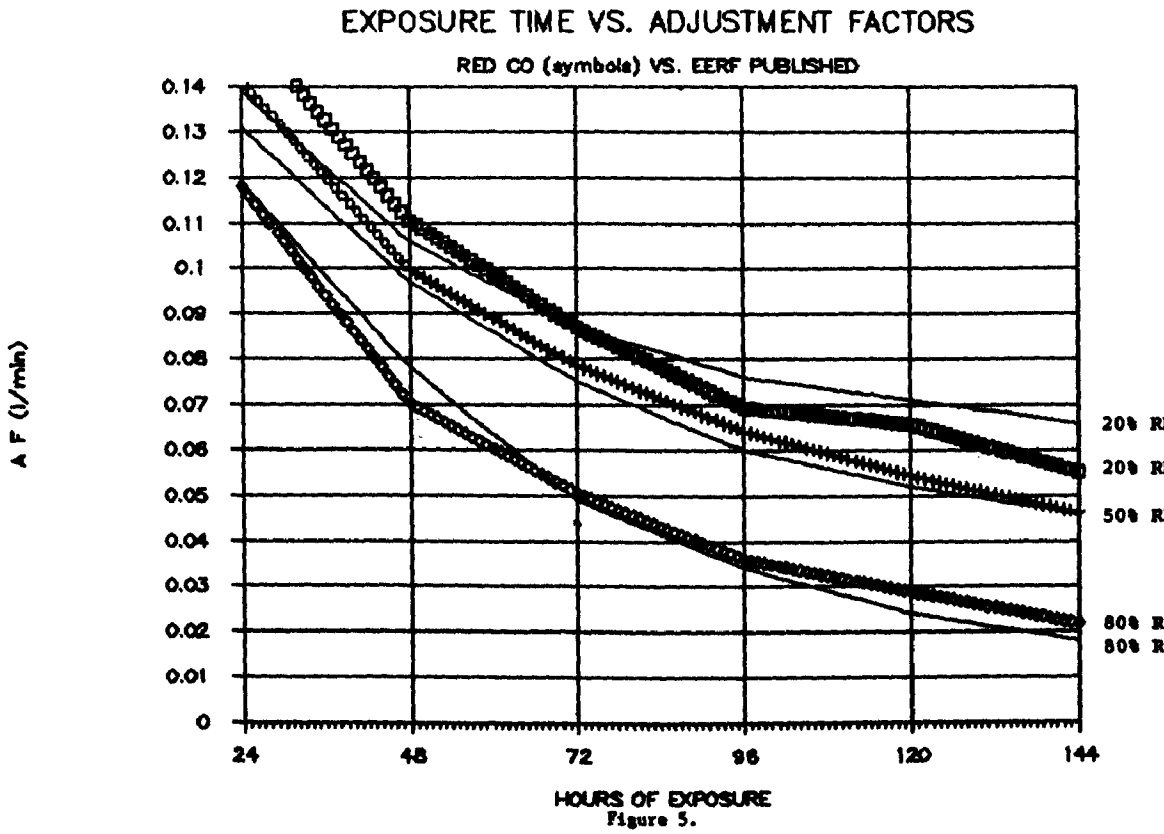

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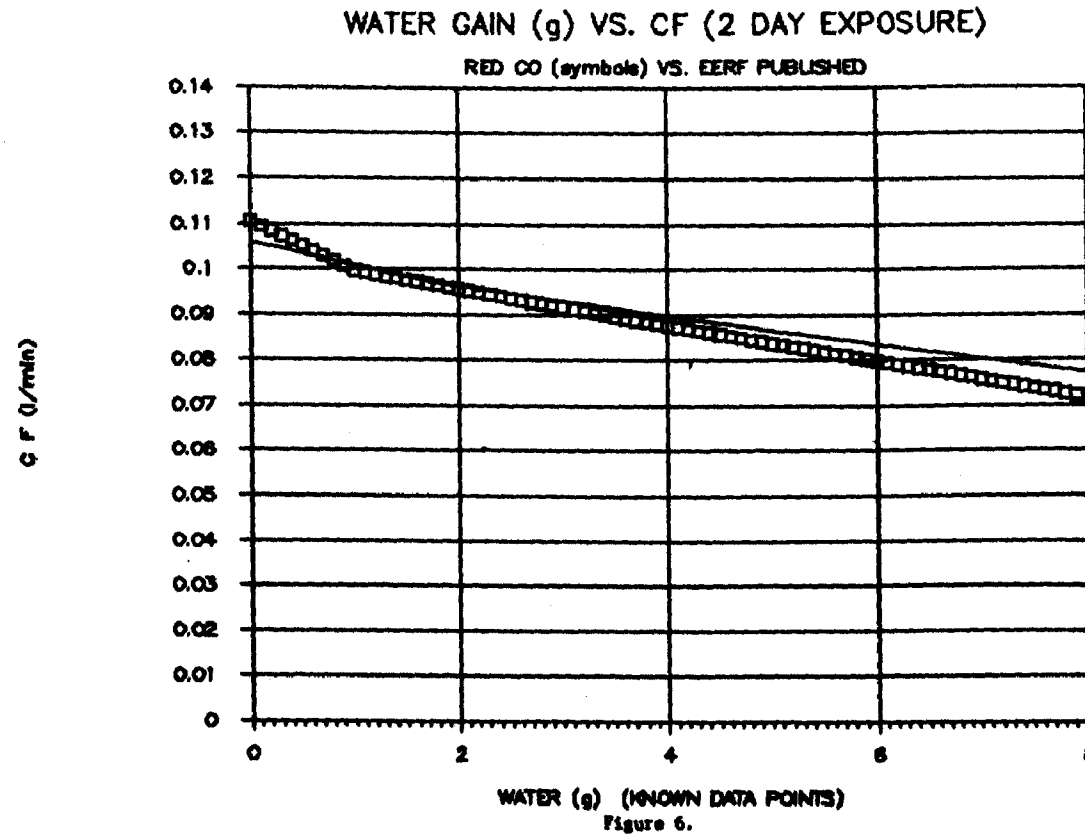

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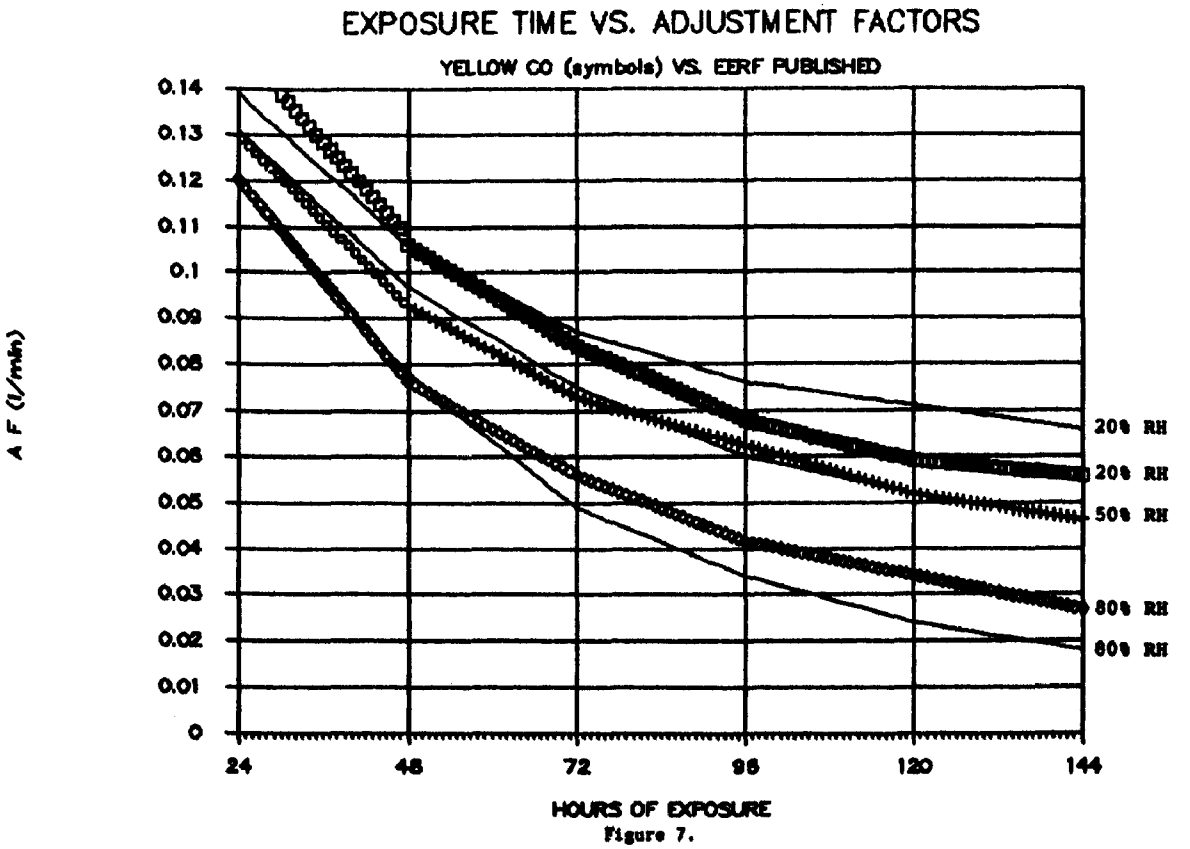

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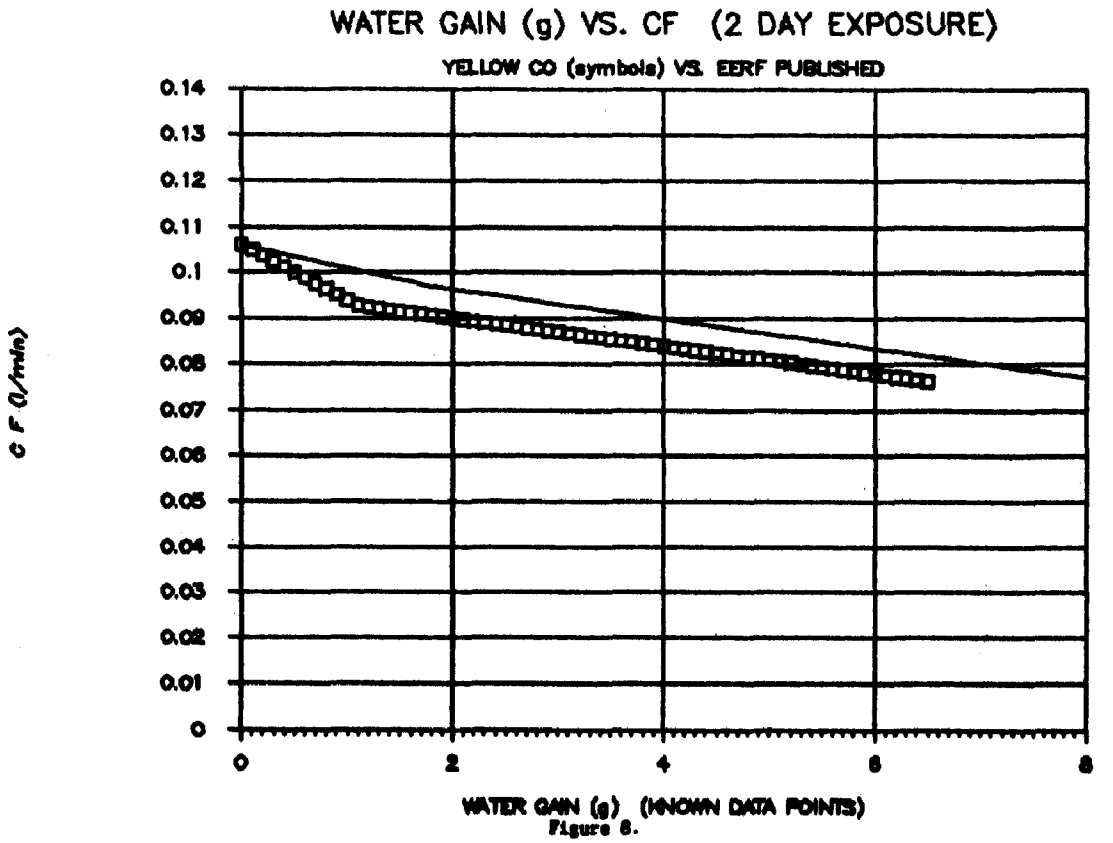

Figure 8.

CALIBRATION OF ALPHA-TRACK MONITORS FOR MEASUREMENT OF THORON (RN-220)

by: Mark D. Pearson*
UNC Geotech
P.O. Box 14000
Grand Junction, CO 81502-5504

ABSTRACT

Several types of commercial alpha-track monitors are supplied in a standard configuration with diffusion barriers that are permeable to both radon (Rn-222) and thoron (Rn-220). These monitors can be modified to exclude thoron by the choice of a diffusion barrier so that the diffusion time across the barrier is on the order of several half-lives of thoron (55.6 sec). Devices with and without thoron diffusion barriers can be simultaneously exposed to determine an appropriate calibration factor for thoron.

Terradex Track Etch® Type M and Type SM monitors, supplied by Terradex with a thoron barrier, and Terradex Radtrak® monitors, modified with a similar thoron barrier, were exposed simultaneously with Track Etch® Type F, Track Etch® Type SF, and Radtrak® monitors in a thoron chamber at the U.S. Department of Energy Grand Junction Projects Office. The thoron barriers prevented thoron from entering the monitors. A calibration equation was determined that permits calculation of integrated thoron exposures from the reported track densities and radon exposures of the simultaneously exposed devices. The thoron calibration factor was established with the greatest precision for the Radtrak® monitors. Study results indicate that Radtrak® monitors provide the greatest sensitivity to thoron.

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INTRODUCTION

The U.S. Department of Energy Grand Junction Projects Office (DOE/GJPO) Radon Laboratory has conducted a number of studies (1,2,3) evaluating the precision and accuracy of alpha-track monitors for the measurement of radon (Rn-222). The GJPO studies and other studies have demonstrated the usefulness of using alpha-track monitors for measurement of integrated radon exposures. Alpha-track devices have also been proposed for estimating exposures of thoron (Rn-220), though few if any studies exist to demonstrate the validity of this approach.

Thoron and thoron-decay progeny are believed to be a significant component of indoor radioactivity. Some researchers have estimated that average dose equivalents due to thoron daughters are 20 to 30 percent of those due to radon daughters (4,5). Several methods exist for the measurement of thoron daughters, but there are no proven, easily applied methods for measuring integrated thoron exposures. Alpha-track devices could provide a simple, inexpensive method for measuring integrated thoron exposures, under appropriate conditions, if a valid procedure for doing so could be demonstrated.

SCOPE

Two types of alpha-track devices are marketed commercially with filters that the manufacturer claims excludes thoron. These devices are the Terradex* Track Etch® Type M and Type SM monitors. Theoretically, these monitors could be used in tandem with the standard Track Etch® Type F and Type SF monitors, which admit both radon and thoron, to determine the net thoron exposure. The net thoron exposure would be calculated from the difference in response between the Type M/Type SM and Type F/Type SF monitors.

Track Etch® Type M, Type F, Type SF, and Type SM alpha-track devices were exposed to a range of thoron and mixed thoron-radon concentrations in air in this study. Additionally, Terradex Radtrak® devices were modified by the replacement of the factory-supplied filter with the thoron-barrier (membrane filter) used in the Track Etch® Type M and Type SM devices. These modified Radtrak® monitors were exposed in conjunction with unmodified Radtrak® monitors.

METHODOLOGY

Thoron (Rn-220) has an extremely short 55.6-second half-life compared to the longer 3.82-day half-life of radon. The Terradex Track Etch® Type M and

*Terradex Corporation, 2 Science Road, Glenwood, Illinois, 60425-1586.

Type SM monitors make use of this short half-life to exclude thoron by employing a membrane with a mean diffusion length that precludes infiltration by thoron. Because of thoron's short half-life, there is a low probability that thoron atoms will cross the membrane. The diffusion of radon into the alpha-track device is only slightly reduced by the increased diffusion length. The standard filters provided by the manufacturer in Track Etch® Type F and Type SF devices and Radtrak® devices readily admit both thoron and radon.

A thoron chamber was constructed at the GJPO Radon Laboratory to conduct exposures of alpha-track devices at constant thoron concentrations and mixed thoron-radon concentrations. The short half-life of thoron requires minimum transit time to produce a relatively constant concentration inside the chamber. If transit time is lengthy, a significant fraction of the thoron entering the chamber will decay before exiting the chamber and a noticeable concentration gradient will exist inside the chamber. The thoron chamber was designed with high air-flow rates and a small chamber volume to overcome this difficulty.

The GJPO thoron chamber is a 20-liter stainless steel cylinder fabricated from a commercial pressure cooker. The chamber dimensions are approximately 30 cm in diameter by 30 cm in height. The top of the chamber is removable, allowing placement of devices in the chamber. Thoron is sampled from one of two ports in the top of the chamber. Room air is pushed through a Pylon Model Th-1025 flow-through thoron source and thence into the chamber. The temperature and relative humidity of the chamber air are continuously monitored but not controlled. The chamber was operated at flow rates ranging from 90 to 500 Lpm for exposures during this study, which produced transit times ranging from 2 to 13 seconds.

The primary thoron measurement is made using a two-filter tube (6). Thoron concentrations are also monitored with two continuous flow-through scintillation monitors. The first monitor samples air from the chamber with as short a path length as possible to minimize the loss of thoron to radioactive decay. The second monitor samples the exhaust of the first monitor with a plumbing system that allows 10 minutes for 99.9 percent of the thoron present to decay prior to entering the second scintillation chamber. This second monitor, therefore, measures only radon.

The thoron concentration at steady-state conditions can be calculated by subtracting the contribution of radon from the counts recorded by the first monitor, and then applying a calibration equation. Hourly thoron concentrations are recorded by a microcomputer-based data acquisition system. The coefficient of variation of hourly thoron concentration readings measured by this method never exceeded 10 percent during any exposure.

Seven exposures of sets of alpha-track devices were conducted in the thoron chamber. The environmental conditions for each exposure are given in Table 1. Six of the exposures were 4 days in duration and one exposure was

TABLE 1. CHAMBER ENVIRONMENTAL CONDITIONS FOR THORON EXPOSURES

Set	Temp (°C)	RH (%)	Duration (days)	Exposure (pCi·d/L) Thoron	Radon
1	36	23	4	1,640	9.2
2	28	50	4	3,516	10.4
3	29	55	4	6,296	9.6
4	32	58	4	5,224	276.0
5	33	46	4	2,916	163.6
6	32	63	4	1,356	164.8
7	31	41	17	21,607	651.1
8	31	41	17	0	651.1

17 days in length. The first three exposures were made in atmospheres containing primarily thoron with little radon. The remaining four exposures were performed in a mixed thoron-radon environment. The 17-day exposure set included a group of monitors that were exposed to the exhaust air from the chamber, with a sufficient time delay to ensure that all the thoron but essentially none of the radon had decayed. This group of monitors, with the time-delay exposure, is Set 8 in the data.

Five Type F and Type M monitors and approximately six Type SF, Type SM, Radtrak®, and modified (equipped with a thoron-barrier) Radtrak® devices were exposed for each test. In most cases, two unexposed monitors were returned for processing along with the exposed devices from each test.

THEORY FOR CALCULATING THORON EXPOSURE

The radon exposure reported by an alpha-track monitor without a thoron barrier cannot be taken at face value as a thoron-exposure measurement because the amount of reported exposure due to thoron and the amount due to radon are unknown. The monitor also has different calibration factors for radon and thoron. Taking the reported exposure at face value as the true thoron exposure would underestimate the actual thoron exposure by 30 to 40 percent in the study's chamber exposures, and would lead to even greater discrepancies as the thoron and radon concentrations became more equal.

A method was developed for calculating thoron exposure from the side-by-side exposure of alpha-track monitors with and without thoron barriers. For monitors without thoron barriers, the method permits the separate calculation of a distinct calibration factor for radon and thoron.

To apply the method, alpha-track monitors with and without thoron-barriers are exposed side-by-side. If the reported radon exposures from each type of

monitor are significantly different, thoron is assumed to be present. The following steps permit calculation of the thoron exposure.

The total track density registered by a monitor without a thoron barrier is the sum of tracks due to the radon and the thoron decay-chain alpha particles. The thoron track density is the difference between the total track density on this monitor and the equivalent radon track density on the monitor. The equivalent radon track density on the monitor without the thoron barrier is determined from the radon exposure reported by the monitor with the thoron barrier. Once the radon and thoron calibration factors have been determined from known exposures, pairs of monitors with and without thoron-barriers can be exposed to unknown mixed concentrations of radon and thoron.

For mixed exposures, the equivalent radon track density on the monitor without the barrier is equal to the radon exposure reported by the monitor with the thoron barrier multiplied by its radon calibration factor.

This value for the equivalent radon track density is then subtracted from the total track density on the monitor without a thoron barrier, to give a value for thoron track density. Finally, the thoron track density is divided by the thoron calibration factor for the non-barrier device, which results in a calculated thoron exposure.

This approach to calculating the thoron exposure can be expressed mathematically as follows. The track density reported on an alpha-track monitor is equal to the sum of the track densities due to radon and thoron. For monitors without thoron barriers (denoted by "f"), and for monitors with thoron barriers (denoted by "m"), the equations are:

$$\rho_f = R_f E_{Rn} + T_f E_{Tn},$$
$$\rho_m = R_m E_{Rn} + T_m E_{Tn},$$

where

ρ_f or ρ_m = track density for f or m device (tracks/mm^2),
R_f or R_m = radon calibration factor for f or m device (tracks/mm^2/pCi·d/L),
E_{Rn} = radon exposure (pCi·d/L),
T_f or T_m = thoron calibration factor for f or m device (tracks/mm^2/pCi·d/L),
E_{Tn} = thoron exposure (pCi·d/L)

These two equations may be simultaneously solved for E_{Tn}:

$$E_{Tn} = \frac{R_f \rho_m - R_m \rho_f}{R_f T_m - R_m T_f}.$$

If the infiltration of thoron through the thoron-barrier is assumed to be zero (which will be shown below to be a reasonable assumption) such that $T_m = 0$, then the equation for the thoron exposure reduces to:

$$E_{Tn} = \left(\rho_f - R_f E_{Rn}\right) / T_f. \quad (1)$$

The term $R_f E_{Rn}$ is simply the equivalent radon track density on the device without a thoron barrier, and the first term is the total track density on the same device; the difference is the net track density due to thoron. The net track density is divided by the thoron calibration factor for the non-barrier device.

A test was performed to evaluate the validity of the assumption that the infiltration of thoron through the thoron-barrier is minimal. Radtrak® monitors with thoron barriers were exposed to a mixed thoron-radon atmosphere, and were also exposed to the same atmosphere after the air had transited a delay volume so that all of the thoron present had decayed prior to reaching the second group of monitors. This second group of Radtrak® monitors with thoron barriers was therefore given the same radon exposure as the mixed exposure group. The reported track densities for both groups should be equal if the thoron barrier absolutely excludes thoron.

The six thoron-barrier monitors exposed to the mixed atmosphere recorded a mean track density of 21.5 tracks/mm^2, with a standard deviation of 1.1. The four thoron-barrier monitors exposed to the radon-only atmosphere recorded a mean track density of 16.5 tracks/mm^2, with a standard deviation of 0.5. Applying a t-test to the comparison of the two means yields a t-statistic of 7.44, which leads to the conclusion that the two means are not equivalent. This result implies that devices exposed to both thoron and radon recorded more alpha tracks than the devices exposed solely to radon. An explanation for this result is that the thoron barrier does not completely exclude thoron from entering the monitor and that the admitted thoron is responsible for the additional tracks.

The total thoron exposure for this test, however, was 21,607 pCi·d/L. Assuming that the 5.0 tracks/mm^2 difference in the two sets of monitors is due to the thoron exposure, the thoron calibration factor for the thoron-barrier devices would be on the order of 5.0 tracks/mm^2 divided by the thoron exposure of 21,607 pCi·d/L, or 0.0002 tracks/mm^2/pCi·d/L. This calibration factor is only 1 percent of the thoron calibration factor for the devices without thoron barriers, or of the radon calibration factor for either type of Radtrak® monitor. Thoron diffusion through the thoron barrier likely appeared in this test due to the large thoron exposure the devices received. In lower thoron exposures that are expected to be more typical, thoron diffusion through the barrier can be ignored for computational purposes with minimal impact.

RESULTS

ANALYTICAL METHOD

Results were analyzed using the track densities reported by Terradex and the measured radon and thoron concentrations in the thoron chamber. A calibration factor for the exposure of the various types of monitors without thoron barriers (Radtrak®, Track Etch® Type F, and Track Etch® Type SF) to a radon-only environment was determined from a least-squares fit through the origin of a set of six or seven exposures. These exposures were conducted separately from the exposures in the thoron chamber and covered a range of 71 to 729 pCi·d/L. The calibration factor determined from these exposures is the radon calibration factor for the monitors without thoron barriers.

Applying the theory, the track density due to radon reported by the monitors without thoron barriers was determined using the radon calibration factor and the measured radon exposure. This track density is the equivalent radon track density for the monitors without thoron barriers. The track density on these monitors due to thoron was then calculated as the difference between the reported track density and the equivalent radon track density.

The calibration factor for the exposure to thoron of monitors without thoron barriers was calculated from a least-squares fit through the origin of the thoron track density versus thoron exposure. The uncertainty in the calibration factor was estimated at the 95 percent confidence level by applying the appropriate factor from the t-distribution table to the error in the slope of the least-squares fit line.

There was an additional uncertainty inherent in the thoron measurement relating to the calibration of the measurement. The "true" thoron concentration value was determined from thoron grab samples taken with a two-filter tube. The two-filter tube measurement was cross-checked with an expected value from the thoron source, based on the stated activity of the source and the flow rate through the source. As a result, there was potentially a systematic uncertainty that would have a bearing on the accuracy of the calibration factor but not on the precision of the calibration factor.

RADTRAK® RESULTS

The exposures of Radtrak® monitors provided results with the greatest precision and the greatest potential for accurate calculation of thoron exposures. The radon calibration factor for Radtrak® monitors without thoron barriers was 0.0266 tracks/mm^2/pCi·d/L. The calculations of average thoron track density for the Radtrak® monitors are summarized in Table 2. Radtrak® monitors were not exposed in Sets 1 and 2.

TABLE 2. AVERAGE TRACK DENSITY CALCULATION FOR RADTRAK® MONITORS

Set	Thoron Exposure (pCi·d/L)	Radon Exposure (pCi·d/L)	Total Track Density (tracks/mm^2)	Equivalent Radon Track Density (tracks/mm^2)	Net (thoron) Track Density (tracks/mm^2)
3	6,296	9.6	125.1	0.3	124.8
4	5,224	276.0	101.0	7.3	93.7
5	2,916	163.6	57.5	4.4	53.1
6	1,356	164.8	30.6	4.4	26.2
7	21,607	651.1	456.6	17.3	439.3

The calibration factor for the exposure to thoron of Radtrak® monitors without thoron barriers, calculated from a least-squares fit through the origin of the average thoron track density versus thoron exposure, was 0.0201 tracks/mm^2/pCi·d/L. Thoron track density is plotted against thoron exposure in Figure 1. At the 95 percent confidence level, the uncertainty in the thoron calibration factor for the Radtrak® monitors was 0.0010 tracks/mm^2/pCi·d/L.

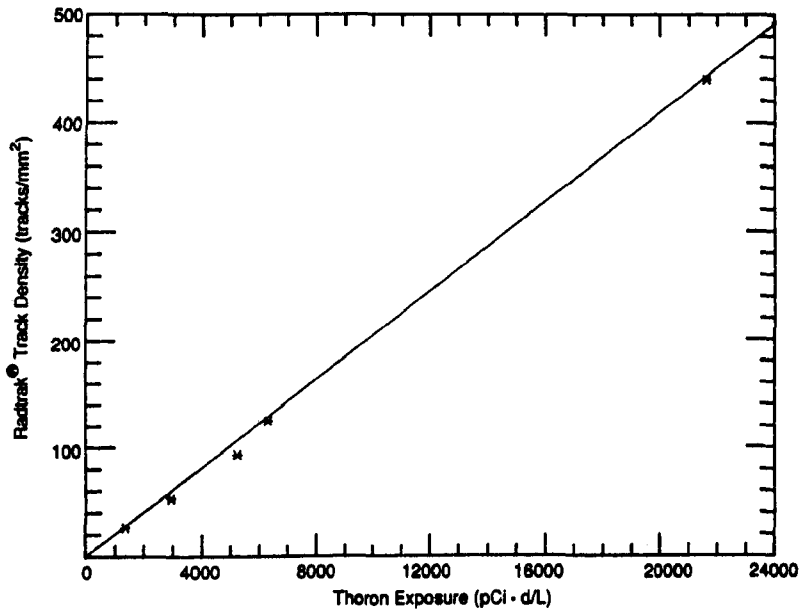

Figure 1. Thoron track density recorded on Radtrak® monitors as a function of thoron exposure.

TRACK ETCH® TYPE SF RESULTS

The radon calibration factor for Track Etch® Type SF monitors (without thoron barriers) was 0.0152 tracks/mm^2/pCi·d/L. The calculations of average thoron track density for the Type SF monitors are summarized in Table 3. The Type SF monitors were not exposed in Set 7.

The calibration factor for the exposure of Type SF monitors to thoron, calculated from a least-squares fit through the origin of the average thoron track density versus thoron exposure, was 0.00940 tracks/mm^2/pCi·d/L. Thoron track density for the Type SF monitors is plotted against thoron exposure in Figure 2. The uncertainty in the least-squares fit at the 95 percent confidence level was 0.00163 tracks/mm^2/pCi·d/L.

TABLE 3. AVERAGE TRACK DENSITY CALCULATIONS FOR TRACK ETCH® TYPE SF MONITORS

Set	Thoron Exposure (pCi·d/L)	Radon Exposure (pCi·d/L)	Total Track Density (tracks/mm^2)	Equivalent Radon Track Density (tracks/mm^2)	Net (thoron) Track Density (tracks/mm^2)
1	1,640	9.2	16.7	0.1	16.6
2	3,516	10.4	41.2	0.2	41.0
3	6,296	9.6	61.2	0.1	61.1
4	5,224	276.0	46.1	4.2	41.9
5	2,916	163.6	29.1	2.5	26.6
6	1,356	164.8	13.8	2.5	11.3

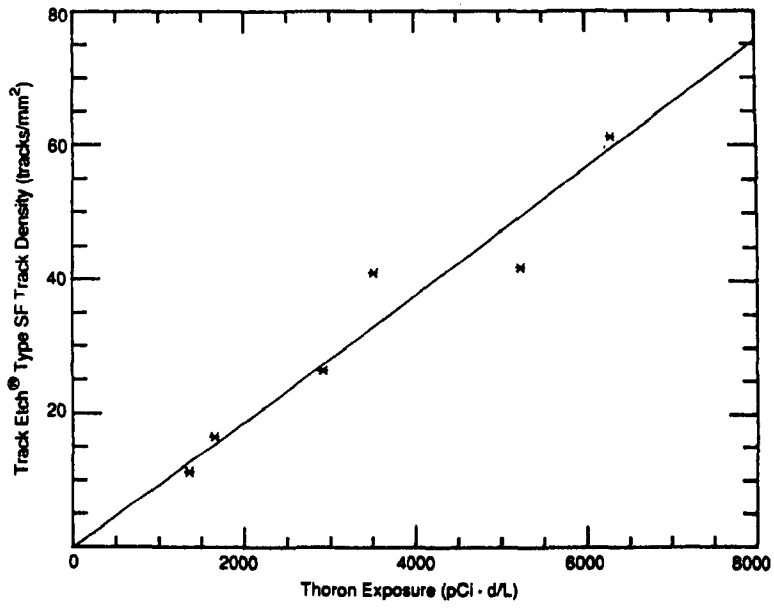

Figure 2. Thoron track density recorded on Track Etch® Type SF monitors as a function of thoron exposure.

TRACK ETCH® TYPE F RESULTS

The radon calibration factor for Track Etch® Type F monitors without thoron barriers was 0.0461 tracks/mm^2/pCi·d/L. The calculations of average thoron track density for the Type F monitors are summarized in Table 4.

The calibration factor for the exposure of Type F monitors to thoron, calculated from a least-squares fit through the origin of the average thoron track density versus thoron exposure, was 0.0133 tracks/mm^2/pCi·d/L. Thoron track density for the Type F monitors is plotted against thoron exposure in Figure 3. At the 95 percent confidence level, the uncertainty in the least-squares fit was 0.0031 tracks/mm^2/pCi·d/L.

TABLE 4. AVERAGE TRACK DENSITY CALCULATIONS FOR TRACK ETCH® TYPE F MONITORS

Set	*Thoron* Exposure (pCi·d/L)	*Radon* Exposure (pCi·d/L)	*Total* Track Density (tracks/mm^2)	*Equivalent Radon* Track Density (tracks/mm^2)	Net (thoron) Track Density (tracks/mm^2)
1	1,640	9.2	41.6	0.4	41.2
2	3,516	10.4	87.3	0.5	86.8
3	6,296	9.6	87.9	0.4	87.5
4	5,224	276.0	101.7	12.7	89.0
5	2,916	163.6	70.9	7.5	63.4
6	1,356	164.8	40.5	7.6	32.9
7	21,607	651.1	298.9	30.0	268.9

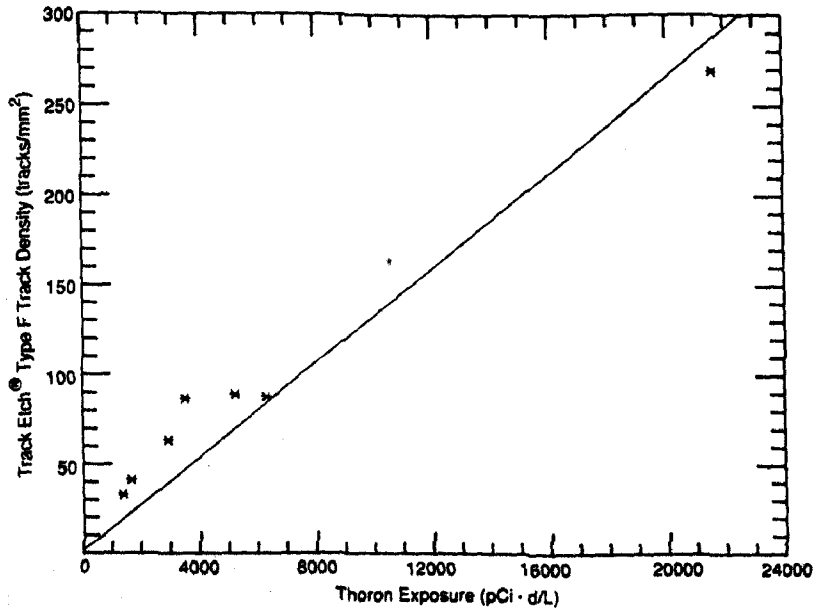

Figure 3. Thoron track density recorded on Track Etch® Type F monitors as a function of thoron exposure.

SAMPLE THORON CALCULATION

Once the calibration factors for radon and thoron for an alpha-track monitor are known, the thoron exposure can be calculated from side-by-side exposures of devices equipped with and without thoron barriers. As an example, consider the Radtrak® monitors with and without the thoron barrier that were exposed in Set 6.

A comparison is made of the reported results from the side-by-side exposure of the monitors. The Radtrak® monitors with a thoron barrier reported 193 pCi·d/L while the monitors without the thoron barrier reported 1,072 pCi·d/L. Since the two types of monitors report significantly different results, the assumption can be made that thoron is present.

Equation (1) is applied:

$$E_{Tn} = \frac{(\rho_f - R_f E_{Rn})}{T_f}.$$

In this case,

ρ_f = 30.6 tracks/mm^2 (as reported by the manufacturer),
R_f = 0.0266 tracks/mm^2/pCi·d/L,
T_f = 0.0201 tracks/mm^2/pCi·d/L,
E_{Rn} = 193 pCi·d/L.

The thoron exposure calculated with Equation (1) is 1,267 pCi·d/L. The measured thoron exposure for Set 6 was 1,356 pCi·d/L. The calculated exposure is within approximately 6 percent of the measured value.

PRECAUTIONS

Alpha-track monitors are known to give occasionally spurious results. In some instances, a monitor receiving a low exposure will report a high result, and the converse can also occur. Since the calculation procedure begins with the comparison between a monitor with a thoron barrier and a monitor without a thoron-barrier, an incorrectly high or low reading will lead to an incorrect conclusion regarding the presence of thoron. To help overcome this inherent difficulty, it is recommended that alpha-track monitors of each type be exposed in duplicates, at a minimum.

While the best results were seen for the Radtrak® monitor with and without the thoron barrier, it should be emphasized that the Radtrak® monitor with a thoron barrier is not commercially available. The devices used for this study were converted by removing the factory-supplied filter from the Radtrak®

monitor and replacing it with a membrane cut from material supplied for use with the Track Etch® Type F monitors.

The measurement of thoron in a structure entails a variety of problems unique to thoron and its short half-life. There may be little thoron present in a structure, or the measured thoron concentration in a room may vary dramatically depending on the location of the monitor and the location of thoron entry into the room due to the short half-life of thoron.

CONCLUSIONS

Alpha-track monitors used in tandem with and without thoron-excluding barriers can provide a reasonable estimate of thoron exposure. The commercially available thoron-barrier does exclude thoron to a significant extent. Thoron exposure can be readily calculated using the track densities and radon exposures reported by alpha-track monitor manufacturers and by applying an empirically determined calibration factor for thoron. Alpha-track monitors are less sensitive to thoron than they are to radon, probably because of thoron's short half-life.

Of the devices evaluated in this study, the Radtrak® monitors provided the greatest precision and sensitivity in estimating thoron exposure. The Track Etch® Type SF monitor provided the least sensitivity for measuring thoron exposure. The Track Etch® Type F monitor provided an intermediate sensitivity for thoron. The diminished sensitivity of the Type F monitor may be due to its large volume that reduces the efficiency by which the short-lived thoron species registers alpha damage tracks on the alpha-track film.

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Quality Assurance Procedures for Home Radon Testing

by: Richard W. Tucker
Radonics Inc.
McLean, VA 22102

John E. McGreevey
Radonics Inc.
McLean, VA 22102

ABSTRACT

Quality Assurance is the critical issue for radon tests that are being made in homes. There are many factors that must be addressed in a positive and methodical manner. These issues include assuring that EPA closed house conditions are maintained; that extraordinary weather conditions did not adversely affect the test; that the equipment is not tampered with or relocated during the test period; that test results are not confused or lost; and, that correct measurement equipment operation was maintained throughout the test. This paper addresses the above issues as well as proposing methods and procedures to assure that reported test results are an accurate indication of the conditions that existed during the test period.

INTRODUCTION AND SCOPE

It is common to read and hear that "radon measurements are easy to make". On the surface this statement may appear to be true. What could be easier than picking-up a test kit while at the grocery store, hanging it in the basement for a few days and sending it off to a lab? If, however, one inquires into the confidence that can be placed in any particular measurement when making health or mitigation decisions based on that measurement, there are many questions that must be answered before the result has any meaning. The elements that surround the process of conducting a radon measurement and reporting the results of the measurement in a manner in which meaning and confidence can be assigned to the results is the process of quality assurance (QA).

Radonics is an engineering firm specializing in radon, especially the making of radon measurements. This paper describes the QA procedures and methods that have evolved over the last three years at Radonics. These procedures have been driven by actual problems encountered in the field and have evolved through and been refined by actual field experience. In fact refinements to the overall QA process are a continuing process which is driven by problems that are still encountered and by the desire to conduct business operations with the highest quality and in the most efficient manner possible.

Radonics performs radon **screening** tests in homes and commercial buildings using sophisticated equipment installed by trained field engineers and technicians. Radon tests can of course be performed in other manners, with other test methods and for other purposes. Radonics, for example, also performs long term radon tests. The differences in the test technology and procedures for the long term tests result in the need for different QA procedures for those tests. This paper is oriented to screening measurements and the QA variations required for other test types are not discussed in this paper. The procedures outlined here, however, should provide a good insight into the types of activities, equipments, procedures and methods that are required to perform a *meaningful* radon test.

MAJOR ISSUES TO ADDRESS

The following list contains some of the issues with which the QA process must ensure conformity during a test cycle:

- Do not confuse, interchange or lose test information (information integrity)
- Insure that EPA closed house conditions are maintained prior to and during the test
- Insure that all other EPA protocols (eg. equipment placement, test times, weather conditions, etc.) are met
- Verify that equipment used is still in calibration period
- Check that equipment used is operating properly
- Analyze all critical test results by multiple sources before reports are released
- Keep all test data for future review and analysis

Fortunately, for Radonics, most of these issues were identified early in the history of the company when test volumes were low enough that each test could receive the personal and intensive attention of one of the principals of the company and mistakes were quickly corrected (although sometimes requiring a re-test). It was easily recognized, however, that if the approach to addressing these above stated issues in a consistent, orderly and as automated a manner as possible was not established, the potential for major problems would grow faster than new business would grow. It was also recognized that, while each of these above issues could be separately treated, many of the issues or at least approaches to addressing these issues were interrelated. A total systems approach would be necessary.

The remainder of this paper discusses each of the above issues and approaches taken to address each of those issues separately and then presents the integrated QA process currently being used that incorporates these approaches. In other words, we will first examine each of the trees and then back-up to examine the forest.

INFORMATION INTEGRITY

Information integrity is the issue dealing with the handling of all of the information associated with a test, from the time that an order is placed and continuing through the entire sequence of events that normally culminates in the issuing of a written report to the purchaser. There are three major areas of mistakes that can degenerate the information content of a test: lost data, data errors and interchanged data. A separate but somewhat related problem is that of protecting access to the data and the QA process.

Lost data may occur because of a number of problems with the data transfer and storage process. Lost data may occur because of manual data handling problems or due to computer failure at a critical time. Lost data problems are alleviated through the elimination of manual data handling, except at the point of order taking, the use of un-interruptable power sources (UPS's) on critical office automation equipment, office data backups every three business hours and storage of data in multiple locations.

The test data collected in the test equipment remains in the test equipment even after the test is complete. The test equipment contains storage media that can hold up to about 75 tests. Each test remains in the test equipment until the storage media fills up with data. At the end of a test, data is transferred from the test equipment to a laptop computer. At the end of each day, data is transmitted from the laptop computer to the office, but the data is not deleted from the laptop computer until several days after the completion of the job. When data arrives in the office over phone lines, two separate copies are placed on two separate servers in the office. On the day after the completion of a job, there are no less than four copies of the test data in existence. If data is lost at any step, it is only required to back-up one step to retrieve the test data.

When the test cycle is totally finished, all of the information files associated with the particular job are compressed into an efficient data storage element and put into a special section of on-line storage and backed-up on tape.

Data Errors occur in the data transfer process when individual pieces of data are incorrectly transferred.

The test sequence usually begins with what remains as the most error prone portion of the test process: the taking of the order. This is usually done over the phone with the person placing the order for the radon test giving data such as address, name of owner, name of person responsible for property access, etc. to a Radonics office worker who must enter this data into a computer terminal. There is a lot of data to be taken and there are many chances of making an error. Critical items are read back to the person placing the order to help minimize errors. Both the person placing the order and Radonics personnel are responsible for errors at this stage.

Once this manual data transfer process in complete, all data transfers are automatic. Those that occur outside of the office computer system (such as machine to laptop and laptop to office exchanges) make use of some form of continuous error checking. Data is transferred over a serial interface from the test equipment to the laptop computer. Every byte that is sent to the laptop is re-transmitted back to the test equipment where it is compared against the data byte just sent. Phone transmissions from laptops in remote locations to the office use cyclic redundancy checks on each 256 byte block to insure error free data transmission.

Interchanged data is perhaps the most dangerous of all possible data problems because the data may appear to have perfect integrity to the QA person reviewing the data. Interchanged data would occur if a field test engineer sent in data that appeared to be taken at one location but in actuality was collected at another.

This problem is guarded against in several ways. Each job is assigned a job number by the office automation system at the time that the order is received in the office. All portions of the test information are tagged with this job number. This number is never assigned to a piece of data through manual means. For example, when an engineer is installing a test system in a home, he does not enter the address or job number with which that data will be tagged. Rather he had received, via modem the previous night, the list of jobs and addresses that he would perform the next day. This information resides on his laptop computer. When he arrives at a job site, the laptop presents him with the list of locations that he is to be at that day. He needs only to use his cursor keys to highlight the current address. This address is double-checked when he returns to pick-up the machine.

The **access to the test data and results** is protected. First, it is protected against certain access by unauthorized office personnel. Only certain individuals are allowed to participate in the in-office QA process. Proper names and passwords must be entered before data can be reviewed and accepted. Results are not available to anyone until the QA process is complete and the results of a test are treated as confidential information. These are given only to the person paying for the test or others designated by the person paying for the test.

These procedures insure that data is correctly transmitted. There is no chance of confusing collected test data with the wrong order. If data is lost at any stage it is backed-up at all previous stages. Access to data is protected. Results are unable to be reported until the QA process is complete, and then only to authorized persons.

EPA CLOSED HOUSE CONDITIONS

Ensuring EPA closed house conditions is perhaps the most important element in conducting screening measurements. The EPA specified conditions provide consistency of measurement results. Without them, screening measurements would be subjected to wide variations in response to changes in living conditions and the effects of the outside environment. Radonics data for 1989 shows that seasonal variations are limited to 1 sigma points that are only 16% from the mean of the annual test average (mean = 2.90pCi/l, 1 Sigma = 0.47pCi/l). Actual variations in a dwelling not under closed house conditions can be over a thousand percent as living conditions and seasons vary.

Other than leaving personnel in a house to monitor the closed house conditions, there is no absolute way to know if EPA closed house conditions have been maintained for the entirety of the test. There are however several means that can be used to test for compliance with closed house conditions. The best and most reliable of these is an analysis of the actual test data.

The measurement equipment used by Radonics logs measurement data, certain environmental parameters and system performance parameters every fifteen minutes. Other than normal diurnal variations and extraordinary weather conditions, variations in the measurement data collected every 15 minutes should remain within certain statistical norms when closed house conditions are maintained. With the absence of EPA recommendations in this area each testing company must define its own limits of acceptability. When test data fall out of the statistical norms the test should be repeated. One of the factors that Radonics considers in the test evaluation part of the QA process is how "well behaved" the test was. Figures 1 through 6 illustrate graphs of data from actual tests.

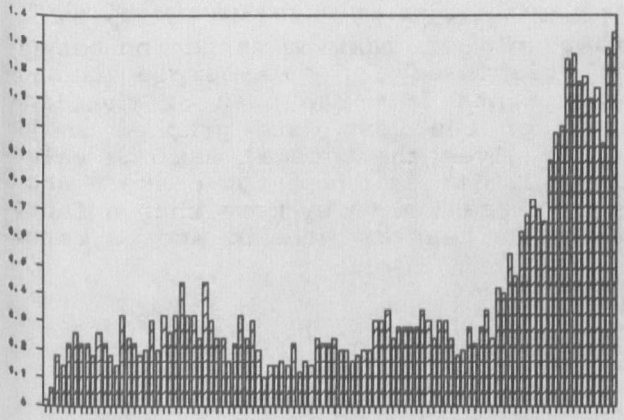

Figure 1 - Test A

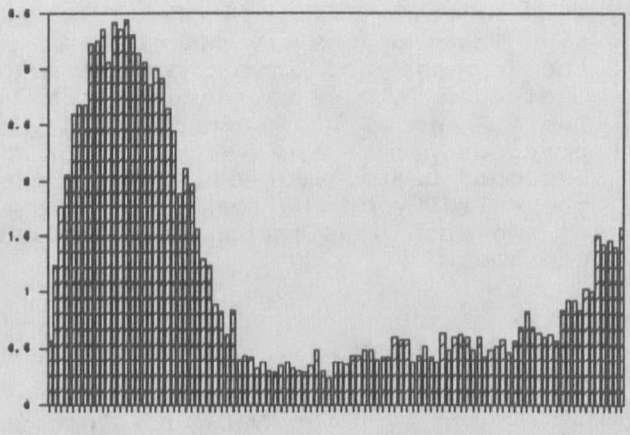

Figure 3 - Test B

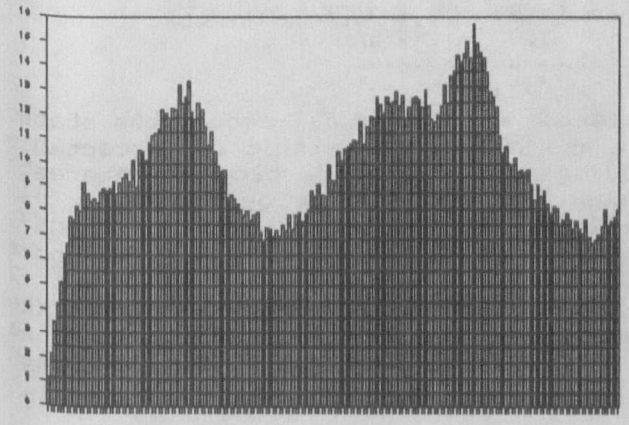

Figure 3 - Test C

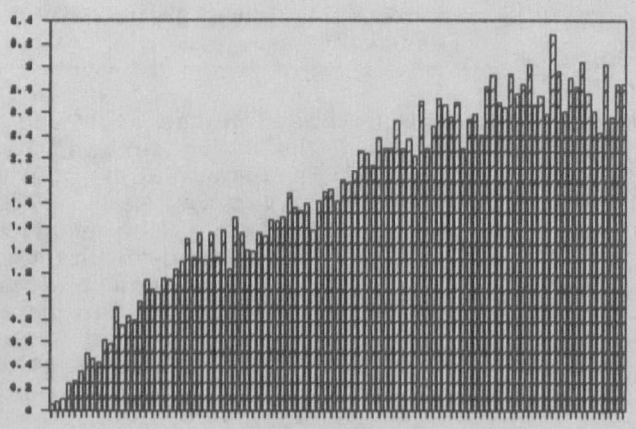

Figure 4 - Test D

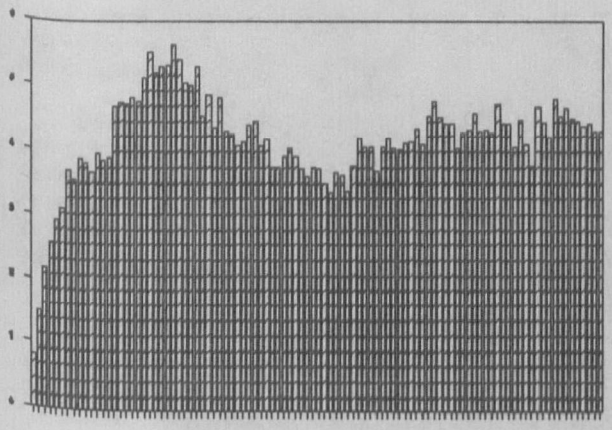

Figure 5 - Test E

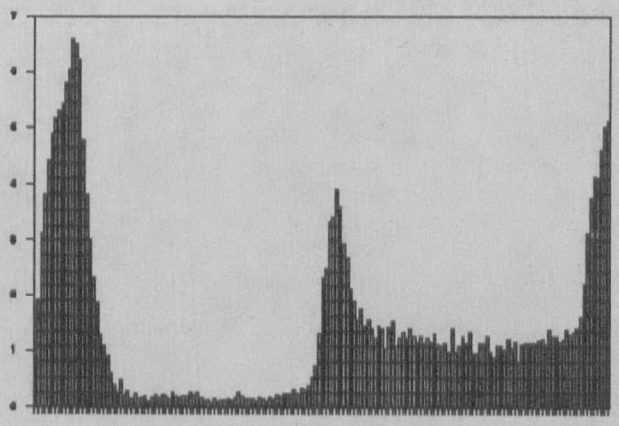

Figure 6 - Test F

Three of the key measurements are ramp-up time, maximum variation between the highest and lowest sample and the "tightness" of the sample data as indicated by the variance of the 1 sigma points from the mean of the data. Table I shows these statistical measures for the test data graphed on the previous page. The last entry in the table gives the typical maximum values for good tests. Exceeding any one of these limits is cause for concern about the validity of the test. Exceeding any of these limits by more than a factor of two should be enough cause to designate the test as invalid and to repeat the test.

Table I - Test Statistics

Test Data	Mean (Av)	1 Sigma Error	Max Ratio	Ramp Up
Test A	0.4 pCi	81 %	11.8	22 hrs
Test B	1.1 pCi	91 %	13.8	2 hrs
Test C	10.1 pCi	19 %	2.3	2 hrs
Test D	2.0 pCi	35 %	7.5	8 hrs
Test E	4.2 pCi	13 %	1.8	2 hrs
Test F	1.3 pCi	109 %	73.8	2 hrs
Typical Max	-	33 %	3.0	2 hrs

The tests labelled Test A, Test B, Test D and Test F all exceed the stated maximum limits. The tests labeled Test C and Test E fall within the acceptable limits. Test F in figure 6 on the previous page is of particular interest. Test F represents a two day test. The house was closed-up prior to the arrival of our test technician. The equipment quickly ramped-up to almost 7 pCi/l during the first two hours of the test. The home owner then opened the doors and windows of the house and the reading fell to outdoor levels (about .2 pCi/l) and stayed low until two hours before the scheduled pick-up time the following day. At the appointed pick-up time we were unable to get back to the property to pick-up the measurement equipment. The home owner then re-opened the house until two hours before our pick-up the next day.

The reasons for the variations in test data are many. They may be occupant interference with the EPA closed house conditions, extreme weather conditions, unusual activity within the dwelling or the phase of the moon. We are usually able to associate variances with an external cause, but not in all cases. The most common problem, by far however, is some sort of interference with the maintenance of the EPA closed house conditions on the part of the occupant. Whatever the reason though, the calculated measured radon (or progeny) average for test data such as that found in tests labeled Test A, Test B, Test D and Test F does not provide meaningful test results and should not be reported.

EPA PROTOCOLS

In addition to closed house conditions, the EPA specifies other test parameters for screening tests. These include directions as to the placement of the measurement device, length of time for which the test must be performed and cautions for unusual weather conditions.

Test personnel should be thoroughly trained in these protocols. To make sure that the equipment has been properly placed, the test engineer or technician sketches on his laptop computer screen, the level of the house on which he has placed the CWLM. He must show walls, windows, crawl spaces, sump pump holes and other pertinent information. On this drawing, he marks the placement of the CWLM and the locations at which grab samples are taken. He is also asked a number of questions by the laptop pertaining to the house construction and conditions. This information is available to the QA personnel for their review.

EQUIPMENT CALIBRATION

It is not left up to the field engineer to make sure that equipment is within the calibration period. In fact, it is usually necessary to pry equipment, that appears to be in good working order, out of the hands of field engineers and technicians.

The last calibration date for each piece of measurement equipment is kept in a data base in the office. The calibration period for most of the equipment used by Radonics is six months. A weekly report is generated that flags equipments approaching the calibration expiration date. These systems are recalled before the expiration date to ensure that no equipment is in the field that has exceeded the calibration date.

CORRECT EQUIPMENT OPERATION

Insuring that machines are within their calibration cycle is a fairly easy thing to do. Just because a measurement system is within its' calibration period however, is no guarantee that it is working properly. It is mandatory that the QA process ensure that constant monitoring of equipment operation is performed.

In home radon testing, measurement equipment is typically moved from a home to a motor vehicle and back to a home every day or two. This process can subject the measurement equipment to shock, vibration and temperature changes, sometimes to extreme levels. This constant movement of equipment can sometimes cause damage. Most precision radon measurement equipment contains components such as solid state alpha detectors and photo-multiplier tubes. These components are the most sensitive of the measurement system components to shock and have the biggest impact on measurement accuracy.

Also the air sampling process required in making precision measurements requires the use of pumps and air paths that must be absolutely air-tight. Pumps and seal are mechanical devices that can fail at any time.

Because of these types of problems, equipment checks every six months or every three months or even every month are inadequate. It is unacceptable to discover three days after test results are reported that a major change in the efficiency of the WL or gas detector has occurred sometime in the last week on the machine that made that test. The test could be repeated, but mitigation or other contractual events may have already occurred.

Because of the possible deleterious effects that can result from an inaccurate test report, the test equipment must be tested immediately prior to and following each test. In fact, it is best if the equipment can test itself during the test. Each type of measurement equipment has its own peculiar weaknesses that must be examined for each test. Radonics primarily uses continuous working level monitors (CWLMs) and the procedures and methodologies that are discussed in this paper address the particular concerns of that type of equipment.

A CWLM has three primary areas that must be monitored to ensure a good test. These are filter integrity, flow and detector efficiency. The filter must not leak. Even pin hole leaks that are invisible or barely visible to the naked eye can cause a large error in the test results and in the direction of yielding a false negative. Air flow and detector efficiency must be exactly known in order to perform an accurate working level (WL) calculation. WL measurement errors are directly proportional to flow errors and detector efficiency errors.

The CWLM equipment used by Radonics tests the filter pressure drop during the equipment power-up built-in-test and then continuously monitors the filter pressure during the test. If the pressure drops below the 8% limit at any time, error flags are set in the data file. The filter pressure is recorded in the data file with the raw measurement data each 15 minute period.

One of the first actions in the CWLM power-up test is a detector check. The equipment operator is requested to insert a radioactive source into the filter holder in place of a piece of filter paper. A two minute count is then performed on the activity of the source. The activity of this source and the number of counts measured in the two minute period are compared against similar data taken at the time of the equipment calibration and stored in the internal system calibration tables. The activity ratio must compare favorably with that of the reference source made at the time of calibration.

Finally, the flow rate must be precisely known. This is typically tested with either a bubble tube device or with a mass flow meter. Care must be taken with a mass flow meter to correct for the altitude, barometric pressure and temperature effects on the mass flow meter. The equipment used by Radonics includes a mass flow sensor, absolute pressure sensor and temperature sensor. The absolute pressure sensor measures the cumulative effect of the altitude and barometric pressure.

These sensors allow for accurate volumetric flow measurements to be made at the initial power-up built-in-test and continually during the test period. The results of the flow measurements are stored with the raw measurement data for each 15 minute period. If internal flow measurement capability is not available, then flow measurements should be made both before and after the test and frequent checks should be performed to demonstrate that the flow for the device is stable and does not "wander" during tests.

In addition to these three critical parameters, there are other system parameters that are important to system operation. These parameters should be measured frequently in the field to ensure that proper system operation continues. An example of another class of parameter that should be closely watched are internal equipment reference, bias and threshold voltages. The Radonics equipment measures these and records the results with the raw measurement data every 15 minutes.

Also parameters that can indicate wear of parts in a measurement system and alert the users to possible up-coming failures prior to the actual failure are important in insuring that tests are properly performed. For example, the equipment used by Radonics measures and records motor current. Motor current has been shown to be a good indicator of pending motor bearing failure.

Even with all of these tests before and during a measurement period, there is still sometimes doubt as to whether or not a test result was correct. It is especially disconcerting to think that a test result might have been a false negative, especially a false negative that was not a few percents off of the mark but rather was substantially lower than the true (high) radon level.

To remove this final concern, the standard QA procedure of making redundant measurements with independent equipment is used. At the initiation and completion of each test at least two WL grab samples are taken. While the accuracy of the grab samples may be substantially less than the accuracy of the CWLM equipment used by Radonics, it does give the QA personnel the assurance that an undetected CWLM error has not occurred which would result in the substantial under-valuation of the actual radon problem.

All of this data must be recorded and transmitted back to the office for QA review. The CWLM equipment used by Radonics stores all of the data discussed in this section (including the grab WL data which is manually entered) and a computer analysis of this data is available to the person performing the in-office QA function.

MULTIPLE INDEPENDENT REVIEW

In any health or safety related decision making process there should be a review process that consists of persons close to the situation and those detached from the situation. There are four principle reviewers of the measurement data in the Radonics QA procedure.

The first review of the measurement data is performed by the measurement equipment itself. The measurement equipment continually measures over a dozen parameters. If all parameters are within normal limits a "GOOD" status is reported. If any one of the parameters varies from within fairly tight limits, a warning or error is reported.

The field engineer reviews all of his tests at the end of the day and records his opinions and observations in the data record before transmitting it in to the office. When the data is received in the office and before it is reviewed by the office QA personnel, the office computer system performs a functional analysis of the test equipment performance and a statistical analysis of the behavior of the measurement data during the test period. The results of all of these analyses and observations are available on the computer screen for the office QA personnel to use in his or her analysis.

In the review of all of this data, the attitude and outlook of the QA reviewer is one of caution. If everything does not appear to be just right, results are not reported and a re-test is performed.

TEST DATA ARCHIVAL

Test data should be kept indefinitely so that any questions that are raised in the future about the validity of the test can be answered. In addition to the test data, all comments made by test and QA personnel should be kept in the computer file.

PUTTING IT ALL TOGETHER

The elements described in the first portion of this paper are only the major elements of the QA process. There are of course dozens of smaller details that have to be addressed to keep tight control over the QA process. A brief summary of the job flow and the QA part in that flow is illustrated in Figure 7 below.

The office system is the glue that ties all of the surrounding aspects of this system together. While the arrows show the flow going from one of the outer blocks to the next for simplicity, each step in this cycle actually begins and ends with an interface to the office system.

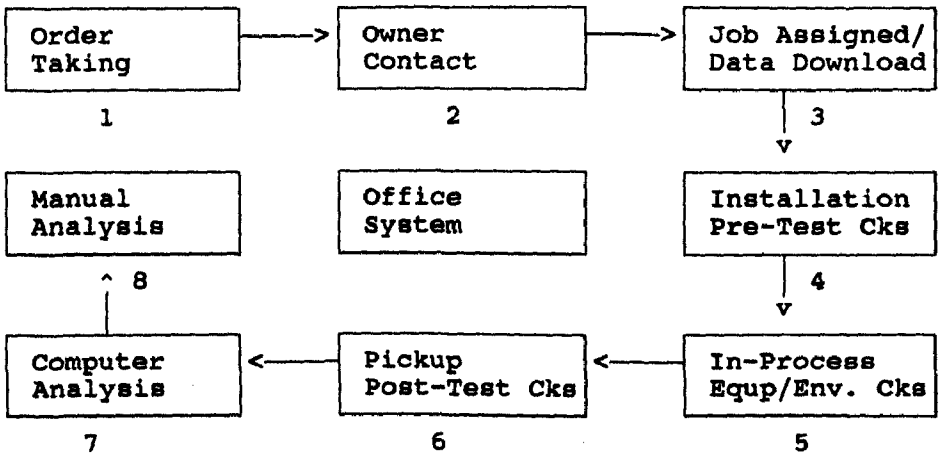

Figure 7.

1. An order is placed. The office personnel must read back critical information to the person placing order to ensure that data has gotten into the computer system correctly.

2. Equipment installation and pickup date/time is established with dwelling occupant. The EPA closed house conditions are explained to the occupant. The 12 hours of closed house conditions necessary prior to the test are especially emphasized.

3. Schedule and job information are sent by modem to test engineer. Automatic error detection and correction techniques are used during computer to computer phone transfer.

4. The test engineer performs equipment installation. The house is checked on arrival for closed house conditions. CWLM equipment is checked and installed. The engineer selects the current address from a menu on his laptop. This address is associated with a job number with which all data for this test will be tagged. Grab samples are taken. Questions are answered and sketch of dwelling made. Closed house conditions are explained and written instructions are left concerning those conditions. A toll-free phone number is left in case of questions.

5. The QA process continues after the engineer leaves as the measurement equipment continues to log radon levels, environmental conditions and equipment operational parameters every fifteen minutes.

6. The test engineer performs equipment pickup. The house is checked on arrival for closed house conditions. The engineer selects the current address from a menu on his laptop. The matching job number is checked against the number selected at installation. At least two more grab samples are taken. The occupant is asked if EPA closed house conditions were maintained and they are asked to sign a document stating that they were maintained. The test data from the CWLM equipment is transferred electronically to the laptop. Data echoing is used to insure 100% data transmission accuracy.

7. All data is sent via modem from the engineers laptop computer to the office. Automatic error detection and correction procedures are used. The data is analyzed by the office computer and any problems are flagged.

8. A QA person having the proper access codes to the QA system, reviews the raw test data and the results of the computer analysis of the test data. If the results are good, a code is selected that allows the release of the report to authorized persons.

CONCLUSION

It is clear from the preceding discussions that while the making of an actual radon measurement may not be difficult, the process surrounding that measurement is very involved. There are many areas that must be given careful and detailed scrutiny. The primary considerations that must be addressed are the maintenance of EPA closed house conditions, the careful monitoring and cross-checking of measurement equipment operation and the process of insuring that results are not confused. Critical pieces of the process that must be present, as a minimum, are a high degree of automation, measurement equipment that records data samples every fifteen to thirty minutes and multiple levels of independent review. All elements of the system must be designed to work together.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

Session III-P2:

Panel Session on Short-/Long-term Radon Measurements

OVERVIEW OF SHORT-/LONG-TERM RADON MEASUREMENTS

by: Richard Sextro
Lawrence Berkeley Laboratory
Berkeley, California

THE RELATIONSHIP BETWEEN WINTERTIME SCREENING AND ANNUAL AVERAGE RADON LEVELS IN U. S. HOMES

by: Melinda Ronca-Battista
Scientific and Commercial Systems Corporation
Falls Church, Virginia

Bonnie Chiles
Dennis Wagner
U. S. Environmental Protection Agency

ABSTRACT

There are a total of about 3700 homes in the country where both short-term, wintertime (screening) and annual average measurements have been made in the same home, and for which the results are available in the published literature. This paper investigates the relationship between these measurements, and presents the percent of homes with different ratios between the screening and annual average concentrations.

The studies vary from a rigorously designed survey of 2000 homes, to small studies of several dozen homes. The methods of measurement include charcoal canisters and alpha-track detectors. Only that data which was from EPA studies was available in non-summary form. An overall pooled frequency distribution was derived by weighting the statistics from each study, and pooling the statistics to form a new distribution. The data are from a large number of homes, from different areas of the country, and when the parameters from the different datasets are compared, they agree surprisingly well with one another.

TEMPORAL PATTERNS OF INDOOR RADON IN NORTH CENTRAL FLORIDA AND COMPARISON OF SHORT-TERM MONITORING TO LONG-TERM AVERAGES

C.E. Roessler, J.W. Revell, and M.J. Wen
University of Florida
Gainesville, FL 32611

ABSTRACT

Both 12-month alpha track (AT) and monthly short-term charcoal canister (CC) measurements of indoor radon-222 (Rn) were made in 37 Gainesville, Florida, houses. Annual average concentrations by AT ranged from 0.5 to 32 pCi/L. Considerable month-to-month variation was observed in the CC measurements; this pattern varied from house to house. The pooled average monthly concentrations showed a seasonal pattern with a winter maximum; a secondary summer maximum was observed for some houses. The annual geometric means of monthly CC measurements correlated well with the year-long AT measurements, but produced values about 12% higher than the AT results.

This study indicates that there are rather large uncertainties associated with estimating long-term Rn concentrations from this form of short-term measurement. For example, a single CC measurement predicted the annual average with a 95% confidence interval of about a factor of 5. This confidence interval is narrower if correction is made for season of measurement and if the estimation is based on the average of more than one short-term measurement.

The data were also evaluated in terms of the probabilities of various long-term Rn levels as a function of the observed short-term measurement.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Since the health risk associated with indoor radon-222 (Rn) is related to the long-term cumulative exposure to the airborne radon decay products, the best way to evaluate indoor radon in a structure is by long-term monitoring under lived-in conditions. On the other hand, there are situations, such as in real estate transactions and in post-construction demonstration of compliance with building codes, with compelling reasons for assessments on the basis of short-term testing.

The precision of a short-term test result as an estimation of the long-term average concentration is affected by systematic variations such as seasonal effects and by random variations associated with the particular device and protocol. Seasonal effects can be induced by environmental variables such as weather and climate as well as by house variables such as construction type and heating, air-conditioning, and ventilation type and operational schedule. These variables in turn may be regionally specific.

This study was prompted by such considerations. The Florida Statewide Radiation Study (1), conducted in 1986-87, indicated evidence for elevated indoor radon in parts of North Central Florida. A winter 1986-87 single-sample screening survey of 67 houses in Gainesville, Florida, found Rn concen-trations exceeding 20 pCi/L in 12% of the houses tested and levels in the range of 4 to 20 pCi/L in an additional 31%.[1] The State of Florida is currently developing a Rn-resistant building code that will have prescriptive (i.e., construction practice) provisions and likely also will have the option for alternative construction if performance criteria are met (i.e., Rn concentrations are acceptable).

In this study, both long- and short-term measurements of indoor Rn concentrations were performed in 37 Gainesville, Florida, houses over a 12-month period from June 1987 through May 1988. The frequency distributions of the measurements were examined, the averages of the short-term measurements were compared to the long-term measurement in each house, and the data were examined for temporal patterns. In addition, an evaluation was made of the ability of various short-term measurement programs to predict annual average concentrations.

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[1]Unpublished, University of Florida.

METHODS

SELECTION OF HOUSES

The houses studied were single-family, detached dwellings. The primary considerations for selection were perceived dependability of the participant, intent to occupy the house for at least a year with no plans for major modification, and construction type. About one-third of the participants were from the participants in the earlier single-sampling screening study, about one-third were city employees, and the balance were other recruits and volunteers. This group of houses is believed to have a bias toward middle income housing. Thirty (81.1%) of the houses were built over concrete slabs, five (13.5%) were over crawlspaces, and two (5.4%) had combination slab/crawlspace substructures. This roughly represents the distribution of substructure types in this region.

MEASUREMENT PROCEDURES

Long-term measurements were made with alpha track (AT) detectors[1] deployed in each house for 12 months and then retrieved and returned to the vendor for processing.

Short-term measurements were made under "lived-in" conditions with charcoal canisters (CCs) deployed for 4 days each month in each house. The participants completed monthly questionnaires in which they recorded whether the house was closed prior to monitoring and also the conditions of occupancy, window status, heating, ventilating, and air conditioning each day during the monitoring period. This information was subsequently used to identify the subset of "closed house" measurements.

The CCs were the 4-in. (10 cm), open-face type.[2] CCs were reused and were regenerated before deployment by overnight heating at 100° C. Counting was performed at the University of Florida using a multichannel gamma spectrometer with a 4 x 4 in. (10 x 10 cm) NaI(Tl) scintillation detector. Radon on the canister was determined by comparison to a standard canister containing a known amount of radium-226. Airborne Rn concentration was calculated by correcting for decay between sampling and counting and by employing an effective sampling rate based on humidity (inferred from weight gain) and length of deployment. Duplicate CCs were deployed in at least three houses each month.

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[1]RADTRAK, Tech/Ops Landauer, Inc., Glenwood, IL.
[2]F & J, Miami, FL.

RESULTS

Of the 37 houses selected, one underwent partial mitigation during the study and ATs from two were reported as damaged. As a result, AT data were available from 35 houses (34 of which were unmitigated for the entire 12 months). CC data were available from 37 houses (36 unmitigated) and CC and AT data pairs were available from 35 houses (34 unmitigated).

A total of 393 CC measurements were recorded; the "closed house" subset consisted of 265 measurements.

Individual short-term concentrations ranged from <0.1 to 110 pCi/L. Arithmetic annual averages of the short-term CC measurements ranged from 0.3 to 43.4 pCi/L for the various houses and geometric means ranged from 0.3 to 35.1 pCi/L. The 12-month AT measurements ranged from 0.5 to 31.9 pCi/L.

FREQUENCY DISTRIBUTIONS

Frequency distributions for the data as collected were highly skewed to the lower concentrations. After making a log transformation, the AT results, the CC data pool, the arithmetic annual house means by CC, and the geometric annual house means by CC all had frequency distributions approximating the normal distribution. Consequently, it was concluded that the data were reasonably described by a log-normal distribution. Thus, data sets were described by geometric means, and statistical testing was performed on log-transformed data. Since there were 12 or fewer CC observations within any individual house, the sample size was too small to determine the frequency distribution of within house values, but it was assumed that these data would also be more closely approximated by a log-normal distribution.

COMPARISON OF AT AND CC ANNUAL AVERAGE CONCENTRATIONS

The CC annual averages by house and AT measurements are compared in the scatter diagram of Figure 1. Comparison of the two data sets by linear regression yielded:

- Using arithmetic mean CC:

  $Y = 0.19 + 1.34\ AT$, $R^2 = 0.95$; and

- Using geometric mean CC:

  $Y = 0.22 + 1.12\ AT$, $R^2 = 0.93$.

The CC results appear to be biased high relative to the AT values by about 0.2 pCi/L. Both forms of CC annual mean correlate well with the AT values ($R^2 > 0.9$), but estimate higher values than the AT method. In addition to the 0.2 pCi/L inter-

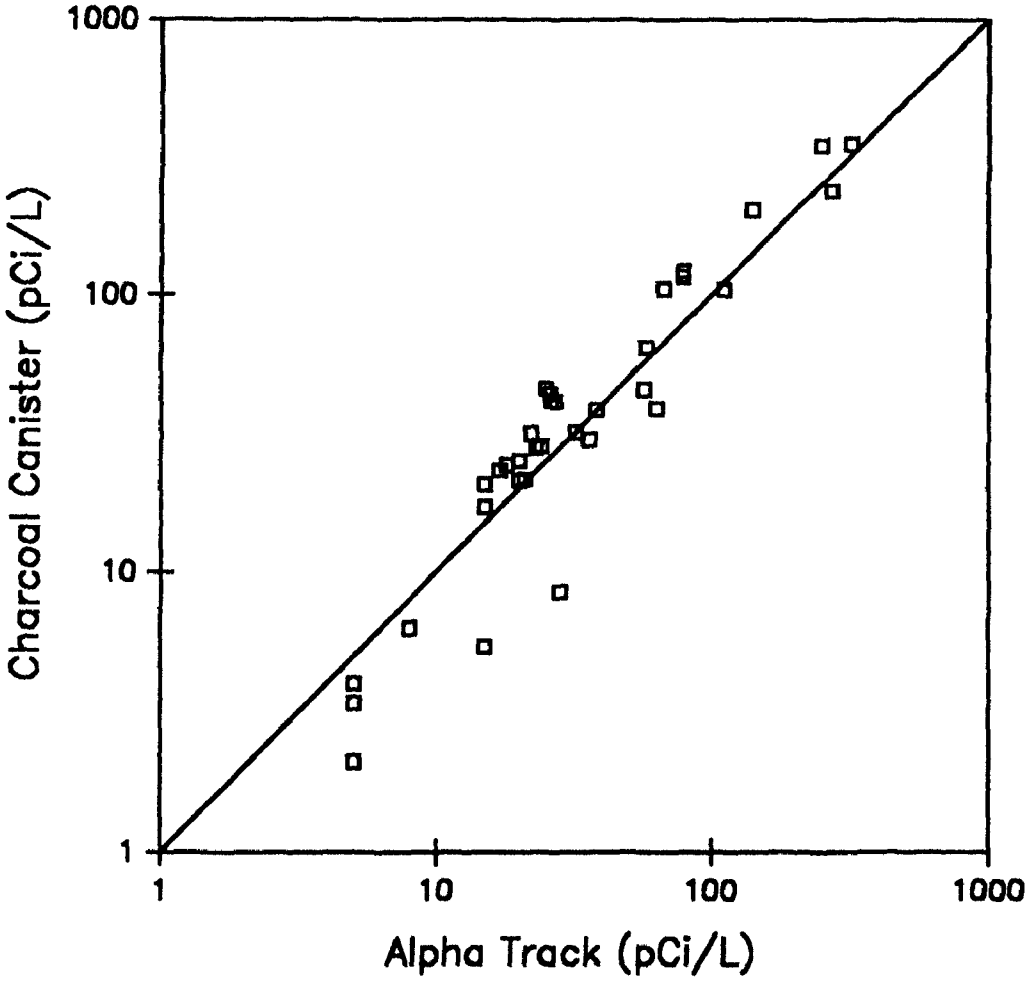

Figure 1. Indoor Radon - Comparison of Geometric Mean of Monthly Canister Measurements with 1-year AT Detectors.

cept, the arithmetic and geometric CC means are higher than the AT values by about 34% and 12%, respectively. This may indicate a difference between the two methods. On the other hand, about a third of the houses were part of an earlier, screening measurement study and the participants may have consciously or subconsciously employed closed-house conditions during CC deployment at times when they would otherwise have had open-house conditions.

TEMPORAL PATTERNS

General Pattern

For each house, the monthly CC values were normalized to the respective CC annual mean (AM). Monthly minimums, geometric means, and maximums of these normalized values are plotted in Figure 2. This figure indicates a general seasonal pattern for the monthly means (MM):

- Winter (Nov-Feb): maximum at MM = 1.6 AM,
- Spring (Apr-May): minimum at MM = 0.8 AM,
- Summer (Jun-Aug): MM = AM, and
- Other months (Mar, Sep-Oct): transitional.

The figure indicates considerable variation between houses. Normalized values as low as 0.6 or less and as high as 2.0 were seen in every month. The maximum normalized concentrations, in addition to having a winter maximum, had a secondary summer maximum; this suggests the possibility of a subset of houses with a summer maximum.

Testing For Seasonal Effect

Months were grouped into equal 3-month quarters for statistical testing. Since it was felt that the use of heating may have a significant effect on pressure differentials and hence Rn entry, and since the heating season in this part of the country generally falls in the months December through February, the seasons were designated:

- Summer: Jun-Aug,
- Winter: Dec-Feb, and
- Fall: Sep-Nov,
- Spring: Mar-May.

The seasonal effect was significant; the following were indicated:

Summer ≤ Spring	<	Fall ≤ Winter
P = 0.11 Not significant @ 0.1 level	Significant @ 0.1 level	P = 0.38 Not significant

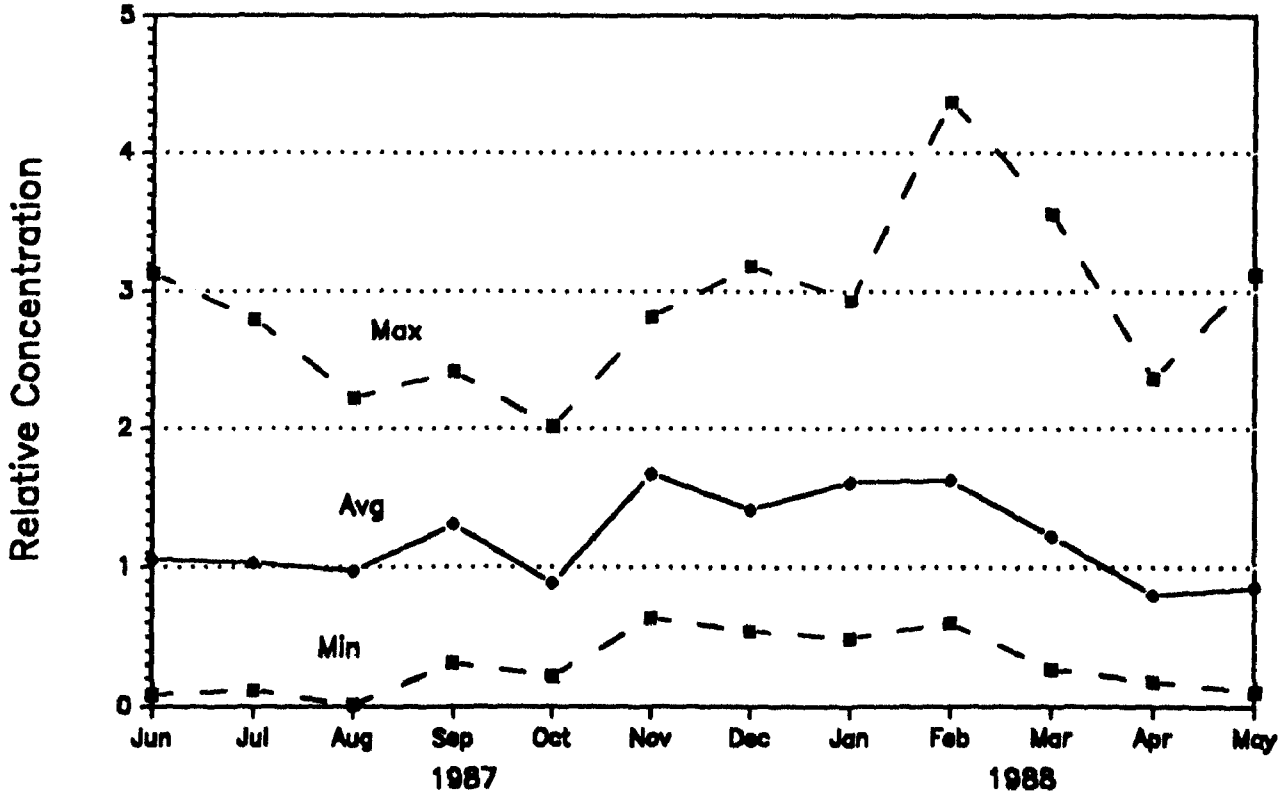

Figure 2. Relative Indoor Radon Concentration By Month (Monthly CC Normalized to Geometric Annual Mean)

The statistical testing confirms the winter maximum suggested by Figure 2. The 3-month spring and fall seasons rank slightly differently in the statistical testing than they did in the Figure 2; months were assigned intutively to equal-length quarters in the former case and based on experience to variable-length seasons in the latter case.

Patterns Within Individual Houses

Inspection of data for individual houses indicates at least four patterns:

1. Some houses had relatively level concentrations with little variation,
2. Some houses had a distinct winter maximum,
3. Some houses had a maximum in the July-August time frame or had a second maximum in the summer, and
4. Some houses had highly variable results without any systematic pattern.

Further inspection of the data for individual houses suggests that some of the seasonal effect may be confounded with ventilation effect (i.e., whether the house was open or closed during the CC deployment). However, this does not appear to be consistently the case; and in analysis of variance, interaction between ventilation effect and season effect was not statistically significant.

ESTIMATION OF THE ANNUAL MEAN FROM CC MEASUREMENTS UNDER LIVED-IN CONDITIONS

The within-house root mean square error for 393 log-transformed CC measurements in 36 houses was 0.825. This translates into a geometric standard deviation of 2.28. If a small number of CC measurements, uncorrected for season of year, were used to estimate CC-derived annual mean Rn concentrations, the 95% confidence intervals would involve multipliers/divisors of 5.0, 3.1, and 2.5 for estimates based on the averages of one, two, or three short-term CC measurements, respectively. Stated another way, the 95% confidence intervals would be:

- Single measurement:
  Lower bound = 0.20 CC, Upper bound = 5.0 CC;
- Average of two measurements:
  Lower bound = 0.32 CC, Upper bound = 3.1 CC; and
- Average of three measurements:
  Lower bound = 0.39 CC, Upper bound = 2.5 CC.

INFERENCES FROM THE CLOSED-HOUSE DATA SUBSET

ESTIMATIONS OF THE LONG-TERM MEAN

The data were analyzed to estimate the constraints associated with the use of a small number of short-term measurements to estimate the long-term average in future testing. The closed-house subset of CC measurements were taken as a representation of what might occur in future short-term testing in this geographic region under closed house protocols. The AT results were taken as the best estimate of the long-term averages in these houses.

For this purpose, the statistic $(Z_j - Y)$ was defined where:

Z_j = Ln CC for the jth measurement[1] and
Y = Ln AT.

The data were fit to the model:

$$Z_{ij} - Y_i = u + a_i + e_{ij},$$

where:
Z_{ij} = Ln CC for the jth measurement in the ith house, Y_i = Ln AT for the ith house,
u = the mean difference between Ln CC and Ln AT measurements,
a_i = the house effect for the ith house, and
e_{ij} = the random error.

For a future short-term measurement, the estimated long-term mean would be:

$Y_i\hat{} = Z_{ij} - u\hat{}$, where $u\hat{}$ = the estimator of u.

In terms of concentration, pCi/L, this back-transforms to:

$$LT(pCi/L) = e^{-u\hat{}}\ C(pCi/L),$$

where LT = the estimated long-term mean and
C = the measured short-term value.

The data were analyzed for the case of the season not specified and also for each of the four seasons. Confidence intervals were calculated for estimations based on a single short-term measurement and also for estimations based on the averages of two and three short-term measurements. The resulting estimation equations and the associated 95% confidence intervals are presented in Table 1.

[1]Note: To avoid having zero values for the log transformation, CC was taken to be the observed value + 0.01 for this analysis.

Table 1. **Predictors for Long-term Indoor Rn Concentrations from Short-term, Closed-house, Charcoal Canister Measurements**

Season of Measurement	Predicted LT Avg.	No. of Measurements	95% Confidence Interval of Predicted LT Avg. Lower Bound	Upper Bound
Not Specified	0.93	1	0.21 LT or 0.20 C	4.72 LT or 4.39 C
		2	0.33 LT or 0.31 C	3.00 LT or 2.79 C
		3	0.41 LT or 0.38 C	2.45 LT or 2.28 C
Spring (Mar,Apr,May)	1.11	1	0.23 LT or 0.26 C	4.34 LT or 4.82 C
		2	0.35 LT or 0.39 C	2.86 LT or 3.17 C
		3	0.42 LT or 0.47 C	2.37 LT or 2.64 C
Summer (Jun,Jul,Aug)	1.22	1	0.23 LT or 0.28 C	4.22 LT or 5.27 C
		2	0.35 LT or 0.43 C	2.84 LT or 3.46 C
		3	0.42 LT or 0.51 C	2.36 LT or 2.88 C
Fall (Sep,Oct,Nov)	0.78	1	0.23 LT or 0.18 C	4.35 LT or 3.39 C
		2	0.35 LT or 0.27 C	2.86 LT or 2.23 C
		3	0.42 LT or 0.33 C	2.38 LT or 1.86 C
Winter (Dec,Jan,Feb)	0.62	1	0.23 LT or 0.14 C	4.33 LT or 2.68 C
		2	0.35 LT or 0.22 C	2.84 LT or 2.76 C
		3	0.42 LT or 0.26 C	2.36 LT or 1.46 C

LT = predicted long-term average

C = measured CC value; either single measuremet or geometric mean

Models:

$LT = a_i C$ where a_i = multiplier for predicting long-term average.

95% Confidence Intervals

Lower bound = $LT / b_i = (a_i / b_i) C$

Upper bound = $LT \times b_i = a_i b_i C$

where b_i = multiplier / divisor for 95% Conf. Interval

PROBABILITIES OF ELEVATED Rn AS A FUNCTION OF OBSERVED MEASUREMENTS

Calculation Of Probabilities

Calculations were made of the probabilities of long-term mean indoor Rn concentrations exceeding various reference values (LT_R). These calculations employed the "t" statistic:

$$t = [Ln(LT_R/C) - u\hat{}]/[Var(Z_j) + Var(u\hat{})]^{0.5}.$$

The probability, P, of a long-term mean concentration, LT, exceeding a stated reference value, LT_R, is the one-tailed probability associated with the "t" statistic; i.e.

$$P(LT > LT_R) = P(-t).$$

The quantity, t, was calculated for observed concentrations and values of LT_R in the range of 1 to 20 pCi/L for measurements made without regard to season and for each of the four specified seasons. The associated probabilities are tabulated in Table 2 for a single short-term measurement. Figure 3 presents a family of curves for the example case of a single short-term measurement with the season not specified. As part of work in progress, figures are being developed for other cases, and probabilities are also being calculated for observations based on the mean of more than one measurement.

Applications Of The Probability Data

These data indicate the consequences of decisions based on various criteria. For example, consider the case where 4 pCi/L is used as the criterion for elevated Rn.

- A program might allow a 25% margin and set the acceptance point for a single screening measurement at 5 pCi/L or less. Table 2 shows that the probability of the true long-term average exceeding 4 pCi/L ranges from 35.5% to 71.6%, depending upon the season of measurement.

- Policy might be considered in which a certain risk of exceeding the primary criterion would be accepted if the probability of exceeding a second, higher level is sufficiently low. For example, for a measurement of 5 pCi/L, the probability of the actual long-term value being >20 pCi/L ranges from <1% to about 6%, depending upon the season.

- Another approach would be to <u>reject</u> a house unless there is strong evidence that the long-term average is not likely to exceed the criterion (i.e., minimize false negatives). For example, to ensure that there is no more than a 10% chance of accepting a house when the long-term concentration actually

Table 2. Probability of Long-term Average Radon Concentration Exceeding Various Levels as a Function of Observed Concentration

Observed Value pCi/L	Probability of LT Average Exceeding Indicated Value				
	1 pCi/L	2 pCi/L	4 pCi/L	8 pCi/L	20 pCi/L
A. Season Not Specified					
1	0.458	0.164	0.032	0.003	<0.001
2	0.777	0.455	0.162	0.032	0.001
3	0.898	0.654	0.317	0.089	0.006
4	0.948	0.776	0.454	0.161	0.016
5	0.972	0.850	0.566	0.239	0.031
6	0.984	0.897	0.653	0.316	0.052
7	0.990	0.928	0.722	0.388	0.075
8	0.994	0.948	0.775	0.453	0.102
9	0.996	0.962	0.817	0.512	0.131
10	0.997	0.972	0.850	0.565	0.161
15	0.999	0.992	0.939	0.750	0.315
20	>0.999	0.997	0.972	0.850	0.453
B. Spring					
1	0.561	0.221	0.046	0.005	<0.001
2	0.857	0.558	0.219	0.045	0.002
3	0.946	0.753	0.406	0.123	0.009
4	0.976	0.857	0.557	0.218	0.023
5	0.989	0.913	0.670	0.314	0.045
6	0.994	0.945	0.752	0.405	0.072
7	0.997	0.964	0.812	0.486	0.105
8	0.998	0.976	0.856	0.556	0.140
9	0.999	0.984	0.889	0.617	0.178
10	0.999	0.988	0.913	0.670	0.217
15	>0.999	0.997	0.971	0.836	0.404
20	>0.999	0.999	0.988	0.913	0.556
C. Summer					
1	0.662	0.260	0.059	0.007	<0.001
2	0.885	0.609	0.258	0.058	0.003
3	0.959	0.793	0.456	0.150	0.012
4	0.983	0.885	0.608	0.257	0.031
5	0.992	0.933	0.716	0.361	0.058
6	0.996	0.959	0.792	0.456	0.091
7	0.998	0.974	0.846	0.538	0.129
8	0.999	0.983	0.884	0.607	0.171
9	0.999	0.988	0.912	0.666	0.213
10	>0.999	0.992	0.932	0.716	0.256
15	>0.999	0.998	0.979	0.867	0.455
20	>0.999	>0.999	0.992	0.932	0.607

Continued ...

Table 2. Continued

Observed Value pCi/L	Probability of LT Average Exceeding Indicated Value				
	1 pCi/L	2 pCi/L	4 pCi/L	8 pCi/L	20 pCi/L
D. Fall					
1	0.373	0.107	0.015	0.001	<0.001
2	0.723	0.371	0.106	0.015	<0.001
3	0.871	0.582	0.237	0.051	0.002
4	0.934	0.722	0.370	0.105	0.007
5	0.964	0.812	0.485	0.169	0.015
6	0.979	0.870	0.582	0.237	0.027
7	0.988	0.908	0.659	0.304	0.042
8	0.992	0.934	0.722	0.369	0.060
9	0.995	0.952	0.772	0.429	0.081
10	0.997	0.964	0.812	0.485	0.105
15	0.999	0.990	0.922	0.692	0.236
20	>0.999	0.997	0.964	0.812	0.369
E. Winter					
1	0.264	0.060	0.007	0.004	0.000
2	0.613	0.262	0.059	0.007	<0.001
3	0.796	0.461	0.153	0.026	<0.001
4	0.886	0.612	0.261	0.059	0.003
5	0.934	0.719	0.365	0.102	0.007
6	0.959	0.795	0.460	0.152	0.013
7	0.974	0.848	0.542	0.206	0.021
8	0.983	0.886	0.611	0.260	0.031
9	0.989	0.914	0.670	0.313	0.044
10	0.992	0.933	0.719	0.365	0.059
15	0.998	0.979	0.869	0.577	0.152
20	>0.999	0.992	0.933	0.719	0.260

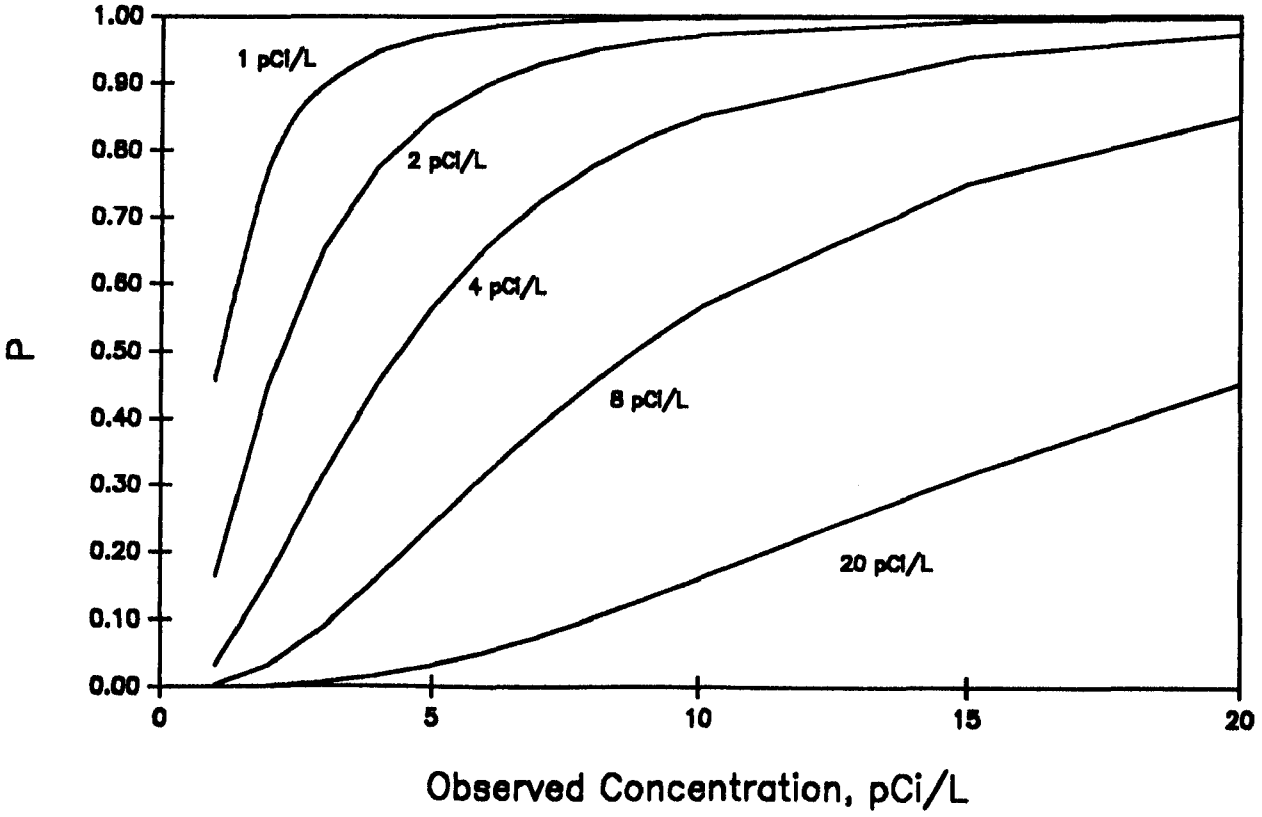

Figure 3. Probability of Long-term Average Rn Concentration Exceeding Various Levels as a Function of Observed Short-term Measurement - Single CC Measurement, Season not Specified.

exceeds 4 pCi/L, the acceptance level must be set at 2.5 pCi/L for winter season measurements.

- An approach at the other extreme would be to pass a house unless there is strong evidence that the long-term average is likely to exceed the criterion (i.e., minimize false positives). For example, to ensure that there is no more than a 10% chance of falsely rejecting a house when the average long-term concentration is actually less than 4 pCi/L, the decision level must be set at 13.5 pCi/L for winter season measurements.

- Another alternative would be a graded approach. For single measurements in the winter season, example criteria might be:

<2.5 pCi/L:	Accept as <4 pCi/L;
2.5 - 13.5 pCi/L:	Perform further testing; and
> 13.5 pCi/L:	Reject as >4 pCi/L.

SUMMARY AND CONCLUSIONS

1. Frequency Distributions of Indoor Rn Concentrations - The AT, CC data pool, and house CC annual mean data sets all had approximately log-normal frequency distributions. Consequently, the data were log-transformed for statistical summarization and testing, and the geometric mean was selected as the most appropriate statistic for summarizing these data.

2. Comparison of CC and AT Measurements - The averages of monthly short-term CC measurements collected over a 12-month period under lived-in conditions correlated well with AT measurements over the same period. The annual geometric means of these CC measurements were generally higher than the AT results - about 12% plus 0.2 pCi/L. The reason for this difference has not yet been determined.

3. Temporal Patterns - In general, for short-term measurements made under lived-in conditions, there was an overall seasonal pattern highest levels in the winter (Nov-Feb), lowest levels in the spring (Apr-May), and intermediate in other months. However, there was considerable variation between houses. Other patterns observed for individual houses included a) a relatively constant concentration, regardless of season, b) a primary or secondary concentration peak in the summer, and c) variability without a discernible pattern.

4. Variability Under Lived-in Conditions - Due to the seasonal effect and other sources of variation, there was a large variance associated with the short-term measurements. The within-house geometric standard deviation for measurement without adjusting for season was 2.28. As a result, the 95% confidence interval on the long-term average concentration as estimated by a single short-term measurement (season not specified) under lived-in conditions is the measured value multiplied and divided by a factor of 5!

5. Estimating Long-term Levels From Short-term, Closed-house Measurements - If the year-long AT measurement is taken as a valid representation of the long-term average Rn concentration, a short term (4-day) closed-house CC test overestimates the LT value by about 7.5% on the average when season of the year is not specified, When season is specified, the average CC measurement ranges from an 18% underestimate during the summer to a 60% overestimate for winter measurements. However, there was considerable variability between houses: in some cases summer measurements overestimated the long-term average.

The single, short-term, closed-house measurement is not a very precise estimate of the long-term average. The 95% confidence interval on this estimate is the estimated value multiplied by and divided by factors on the order of 4 to 5, depending upon the season of measurement.

6. Testing for Compliance With a Performance Standard - Data presented in this paper provide a means for evaluating the uncertainties associated with decisions based on short-term testing and various decision levels.

7. Further Needs - In specifying pass/fail criteria for testing, serious thought will have to be given to determining the acceptable level of error. It also is important to determine the extent to which the results of this study are generally applicable to other years, other geographic areas, and other short-term methods. There is an urgent need for measurement methods and protocols with lower variances than observed in this study.

ACKNOWLEDGEMENTS

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STUDY ON THE RELIABILITY OF SHORT-TERM MEASUREMENTS TO PREDICT LONG-TERM BASEMENT RADON LEVELS IN A RESIDENCE

David A. Hull and T. Agami Reddy
Center for Energy and Environmental Studies
Princeton University
Princeton, NJ 08544-5263

ABSTRACT

This paper addresses the statistical issues relating to the reliability of short-term measurements (during 1 -, 2 -, 4 -, and 7-day measurement periods) for predicting long-term radon levels in a residence. The study is based on measured continuous daily-averaged basement radon data for an entire year, for a residence in Princeton, New Jersey. Day-to-day variability, both during individual months and over the different seasons of the year, is quantified statistically. The practical implication of the resulting summary statistics is that, in the absence of detailed seasonal knowledge, it seems impossible to make accurate long range predictions by means of observations in a single season. Quantile-quantile plots for each of the four different averaging periods and for different seasons of the year are generated in order to test the probability distributions for both normality and log-normality. Finally, a short-term measurement strategy which involves performing measurements during each of the four seasons of the year is studied and its superiority over a single short-term measurement is quantified in probabilistic terms.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.[1]

1. STATEMENT OF PROBLEM

There is increasing concern over the quality and validity of short-term radon measurements in residences as accurate predictors of long-term radiation exposure. It is well known (for example [1,2]) that indoor radon concentrations vary widely from day to day and that they often show strong seasonal patterns. Unfortunately, these patterns may be considerably different from house to house, depending on the soil type and permeability, the climatic parameters, the house construction, the house dynamics, and the occupant behavior which can be to some extent stochastic. Therefore, before

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any concrete standards on radon exposure can be formulated, it is important to develop short-term screening measurement techniques which would enable reliable estimation of the long-term average. Unfortunately, such a study would require taking accurate long-term measurements in a large number of different homes all over the country, an expensive and time-consuming undertaking. Consequently, it is not surprising that there are only a few studies of this type in the published literature [3-6]. The objectives of the present study, though limited to a single residence and to a single year, will be to quantify and study the variability of indoor radon over the year and to evaluate a new screening technique involving four seasonal short-term measurements in terms of its ability to yield predictions closer to the long-term average.

2. DESCRIPTION OF DATA

Princeton University has collected continuous radon measurements during a full year in an unmitigated home in New Jersey (which we shall refer to as H22). The radon concentration in the basement of H22 was measured by a Wrenn chamber every 6 seconds and the data were stored as 1/2-hour averages. For the purpose of this analysis, the data set was reduced further by creating daily averages. The overall average radon concentration for the year (March '88 to Feb '89) was 41.5 pCi/L with a standard deviation of 18.7 pCi/L. Thus, the daily mean basement radon levels in H22 show significant variation on a day-to-day basis. In order to measure the variation over the year, the data were initially broken down into monthly averages (see Table 1 and Fig. 1). They reveal a definite seasonal pattern, with the basement radon concentration dropping to roughly half of the yearly average for 2 months during the summer (July and August) and climbing well above the 40 pCi/L mark during the fall. We also note that it is precisely these 2 months in summer which have coefficient of variation (CV) values twice or even three times those of other months. We attribute this fall in average radon values (and thereby an increase in CV values) to increased air-conditioner (HAC) operation due to an unusually hot summer. It must be stated that this is a house-specific phenomenon since the opposite was observed in another house [7].

From Table 1, we note that the winter months (December to April) have monthly mean radon values closest to the yearly mean, while the CV values are lowest. In fact, the standard deviations during December and January are seen to be about half of those during the summer months and the fall seasons. This suggests that, if single short-term measurements have to be done, it is advisable to do so during these months (especially December and January). Since this observation is house specific, the single most important implication of the summary statistics given in Table 1 is that, in the absence of detailed seasonal dynamics of the particular house, it seems impossible to make accurate long-term radon predictions by means of observations in a single season.

Next, we chose to divide the year into five distinct seasonal groups (Early Spring, Late Spring, Summer, Fall, and Winter) which correspond roughly, but not exactly, to the seasons after which they are named (see Table 2). The clear seasonal difference indicates that it would be more meaningful to compare various short-term vs. long-term averages for each of

these five periods instead of simply considering the entire year. By proceeding thus, which is akin to a stratified sampling approach, one can hope to decrease the error bounds. We note again that Winter and Spring seem to be the seasons with mean basement radon levels closest to the annual average and with lowest CV.

3. ANALYSIS AND DISCUSSION

3.1 BOX AND WHISKER PLOTS

The goal will be to compare the ability of daily-average basement radon measurements and those of 2-day, 4-day, and 7-day moving averages to predict the overall seasonal average. Box and whisker plots of the data are presented in Fig. 2. The dashed line represents the seasonal average, while the median is shown as a solid line. The ends of the box are referred to as the "hinges" and designate the lower and upper quantiles. Points that are 1.5 times the interquantile range are called the "inner fences." The "whiskers" drawn from each hinge extend to the most extreme observation inside the inner fence. Outlier points are shown individually. The standard deviations for each period are given in Table 3. As expected, the standard deviation decreases with a longer measurement period. However, the number of outliers often does not, although their magnitude certainly decreases. Since radon often follows periodic cycles that are several days in length (i.e., a positive autocorrelation coefficient is present [7]), 2- and 4-day averages often result in less smoothing of the data than would be expected. In other words, if the radon value is high on a specific date, it is quite likely that the concentration would also be high on the previous day and on the day after, resulting in relatively high values even when 2- or 4-day averages are taken.

3.2 STATIONARY DISTRIBUTIONS

The next topic which has been addressed relates to the shape of the stationary distribution of these averages. Larsen [8] found that concentrations of air pollutants are log-normally distributed for all averaging times. However, a recent study [6] stated that the radon probability distributions looked more normal than log-normal. In order to test the above claims, the data were plotted against both a normal and a log-normal distribution using quantile-quantile plots (Q-Q plots) (see any appropriate statistics book, for example [9]). The sorted values of the daily averaged radon data (or their natural logarithms) are plotted on the Y-axis of the Q-Q plots, while the corresponding quantiles of the normal distribution (or of the log-normal distribution) are plotted on the X-axis. Thus these Q-Q plots give good insight into the form of the distribution since the more linear the point spread, the more closely do the data approximate the corresponding probability distribution.

These plots are shown in Figs. 3 (a and b) for summer and winter seasons and for the 1 -, 2 -, 4 - and 7-day averages. For the 1-day averages, it appears that the log-normal provides a better approximation than the normal distribution. However, for Late Spring and Summer, it appears that the true distribution of the data might be somewhere in-

between. As the averaging time becomes longer the linearity of the data seems to deteriorate, indicating that longer-term moving averages deviate from both the normal and the log-normal distributions. From the quantile plots, it seems reasonable to fit all the data using the log-normal distribution. Although it fits some seasons and averaging periods better than others, it is probably better to use only one distribution in order to maintain consistency. Given the limited nature of this study, we have refrained from performing stricter statistical tests on these probability distributions. Future studies involving more houses and locations should address this aspect in a more detailed manner.

3.3 QUANTIFYING SHORT-TERM MEASUREMENT RELIABILITY

The final stage in the analysis will be to quantify the reliability of the short-term measurements in a particular season relative to the long-term annual average. The measure of reliability will be quantified by the probability that a random observation falls within a certain range about the true annual average. Based on the true approximate yearly average of 40 pCi/L, the following ranges have been chosen in the subsequent analysis:

Radon Range (pCi/L)	Magnitude of Error (%)
35-45	12.5
30-50	25
25-55	37.5

The results for each season and each averaging period are presented in Table 4. The numbers in the table represent the percentage probability that an observation taken from the theoretical log-normal distribution and fitted to the actual data will be within the specified range. In other words, the fractional area of the theoretical log-normal probability distribution of that particular season and averaging period contained between the specified radon level range is equal to the number cited in Table 4. For example, a basement radon level chosen from the log-normal distribution fitted to the Late Spring 1-day averages would have a 44% chance of being between 30 and 50 pCi/L. For better visualization of the numbers in Table 4, we present Fig. 4 to illustrate variation of probability values at the 25% uncertainty level with averaging period during different seasons.

From Table 4 and Fig. 4, we note that during Winter and Spring, the chances of accurate prediction improve as the measurement period increases in length. For the Summer and Fall, since the seasonal average is very different from the yearly average, the probability of a short-term test result falling within the allowable range is poor, regardless of the averaging interval. Therefore, a single short-term measurement is likely to be accurate only if it is measured during the right season. Even in the best of situations (i.e., during Early and Late Spring and Winter) the probability of a 7-day average test result being within 12.5% of the yearly average is still only about 45-60%. Clearly, in H22, it is impossible to use a single short-term measurement to estimate the annual long-term average to within 12.5% of the actual true value.

3.4 AN ALTERNATIVE SHORT-TERM MEASUREMENT STRATEGY

A previous study [4] suggested that an average of measurements taken during two different seasons (Summer and Winter, for example) would provide a more satisfactory estimate. In H22, this method would probably provide better results, but it still has the problem that, unless the proper two seasons are chosen, the results may still be poor. For example, an average winter measurement will be quite close to the yearly average, 39.9 vs. 41.5 pCi/L. However, if this were combined with an average summer measurement, the resulting estimate would be (39.9+24.1)/2 = 32.0 vs 41.5 pCi/L, which is considerably worse.

In this study, we have chosen to investigate an alternative strategy whereby one takes one short-term measurement during each season of the year. In order to test this strategy, four random samples of 1000 observations were generated from each of the theoretical distributions of the Late Spring, Summer, Fall, and Winter seasons. Sampling without replacement was then done from each of these four seasonal samples to form 1000 sets of four different short-term estimates. An analysis of this data produced the results in Table 5 which are illustrated in Fig. 5. This prediction method clearly outperforms a single short-term average as is obvious from Figs. 4 and 5. It is important to note that, even using 7-day averages, the chance of getting a measurement within 10-15% of the yearly average is only 68%. However, if a measurement that is accurate to within 25% is acceptable, then the probability leaps upward to 96%. This means that, from four seasonal 7-day short-term estimates, the true annual average radon value of this particular house can be predicted to within bounds of ± 25% at a confidence level of 96%.

4. CONCLUDING REMARKS

It is important to realize that the above conclusions are based on the analysis of only a single residence. However, some of these conclusions can certainly be generalized. The combination of high day-to-day variations with strong seasonal fluctuations in radon concentrations makes it very difficult to reliably estimate the yearly average using a single short-term measurement. Increasing the measurement length improves the accuracy of a seasonal estimate, but it does not guarantee that the seasonal average accurately reflects the yearly average. Taking a measurement from each season appears to provide the best results. Though this strategy is perhaps not practical, the analysis based on such a strategy does provide us with an upper bound to the prediction accuracy one could realistically achieve from random short-term radon measurements made in a house: i.e., without explicit recognition being given to the physical forces which influence indoor radon levels.

In addition, the calculations in this study did not include instrument error, which ranges from 10 to 15% for the Wrenn chamber used for collecting data. Other short-term instruments like charcoal canisters are likely to be even less reliable. This means that, realistically, the highest accuracy one could hope to achieve in terms of predicting annual average indoor radon levels from short-term measurements is likely to be only about 50%. Not only should similar studies be conducted in more houses and in other geographic locations in order to verify the conclusions presented here, but

year to year variation in the same house should equally be addressed in the framework of future studies.

5. ACKNOWLEDGEMENTS

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Table 1 - Monthly basement radon statistics for H22

Month, 1988-1989	Days of missing data	Monthly radon average (pCi/L)	Standard deviation	Monthly / Yearly	Coefficient of variation
March	10	48.7	12.2	1.17	0.25
April	7	39.8	10.4	0.96	0.26
May	0	46.9	20.1	1.13	0.43
June	0	42.3	16.2	1.02	0.38
July	0	21.0	14.9	0.51	0.71
August	0	27.2	19.9	0.66	0.73
September	0	57.1	17.8	1.38	0.31
October	0	56.8	20.2	1.37	0.36
November	0	37.9	13.3	0.91	0.35
December	7	40.0	9.3	0.96	0.23
January	4	40.7	9.6	0.98	0.24
February	0	41.3	15.5	1.00	0.38
Year	28 of 365	41.5	18.7	1.00	0.45

Table 2 - Seasonal basement radon statistics

Season	Months of year	Days during which data are available	Seasonal radon average (pCi/L)	Standard deviation	Monthly / Yearly
Early Spring	3-4	44 of 61	44.0	12.0	1.06
Late Spring	5-6	61 of 61	44.6	18.3	1.07
Summer	7-8	62 of 62	21.0	17.7	0.58
Fall	9-10	61 of 61	57.0	18.9	1.37
Winter	11-12, 1-2	109 of 120	39.9	12.2	0.96
Year	3-12, 1-2	337 of 365	41.5	18.7	1.00

Table 3 - Standard deviations of basement radon averages for different averaging periods

Averaging Period	Early Spring	Late Spring	Summer	Fall	Winter
1-day	12.0	18.3	17.7	18.9	12.2
2-day	9.8	16.2	15.2	15.8	11.2
4-day	7.6	13.4	13.1	12.8	10.1
7-day	6.1	10.1	10.8	10.7	9.4

Table 4 - Reliability of short-term vs. long-term measurements for different averaging periods and different seasons for H22 whose yearly basement radon level is 41.5 pCi/L. Numbers in the table represent the percentage probability that an observation taken from the theoretical log-normal distribution and fitted to the actual data will be within the radon range cited.

Range of average radon (pCi/L)	Early Spring				Late Spring				Summer			
	1d	2d	4d	7d	1d	2d	4d	7d	1d	2d	4d	7d
35-45	36	43	50	57	22	27	33	45	07	07	07	05
30-50	64	73	81	87	44	52	61	76	15	16	15	12
25-55	82	88	93	97	62	71	80	91	25	25	25	23

Range of average radon (pCi/L)	Fall				Winter			
	1d	2d	4d	7d	1d	2d	4d	7d
35-45	19	20	18	15	32	34	37	39
30-50	37	37	36	33	60	64	67	71
25-55	51	52	53	54	81	84	87	90

Table 5 - Reliability of short-term vs. long-term measurements for different averaging periods when a mean value of four seasonal measurements is taken. (For H22, the yearly basement radon level is 41.5 pCi/L.)

Range of average radon (pCi/L)	Magnitude of uncertainty (%)	Averaging period			
		1d	2d	4d	7d
35-45	12.5	41	51	57	68
30-50	25	74	82	89	96
25-55	37.5	89	95	98	99.7

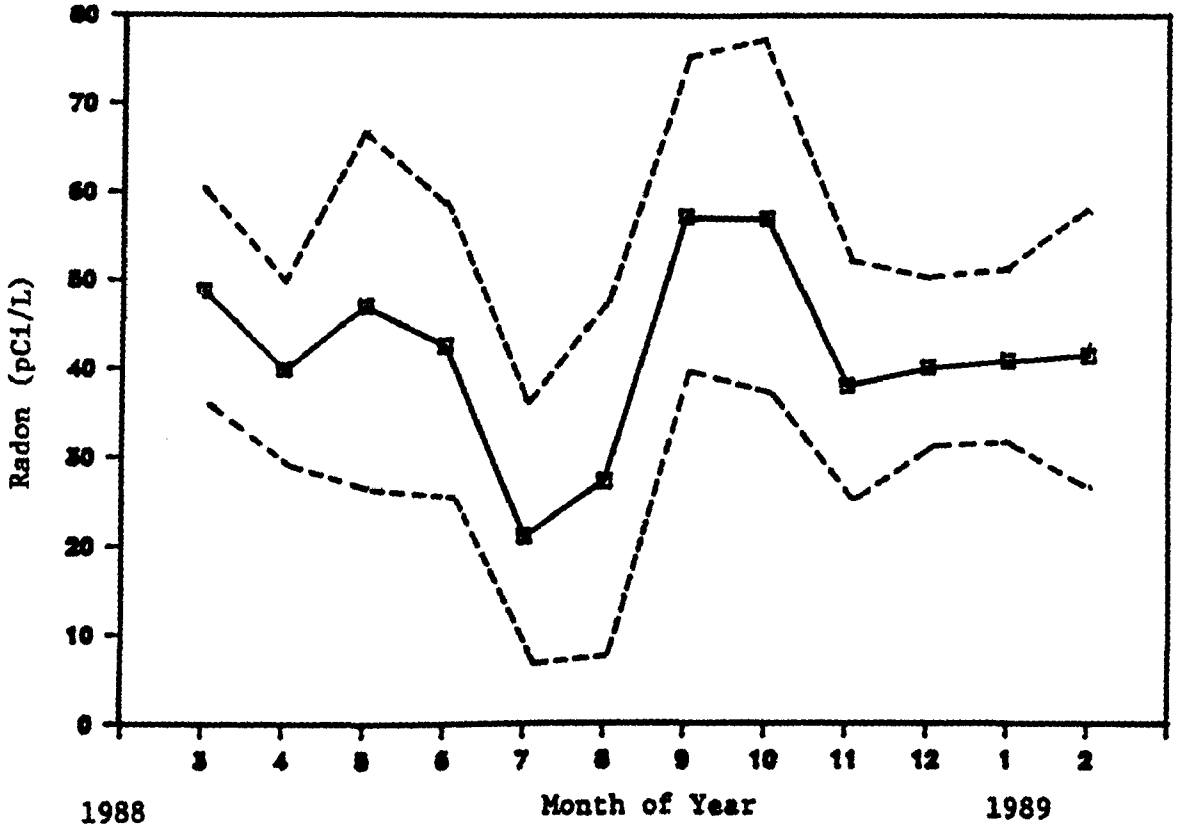

Figure 1 Month by month variation of the monthly mean daily average basement radon for H22. The dashed lines represent ± 1 standard deviation range.

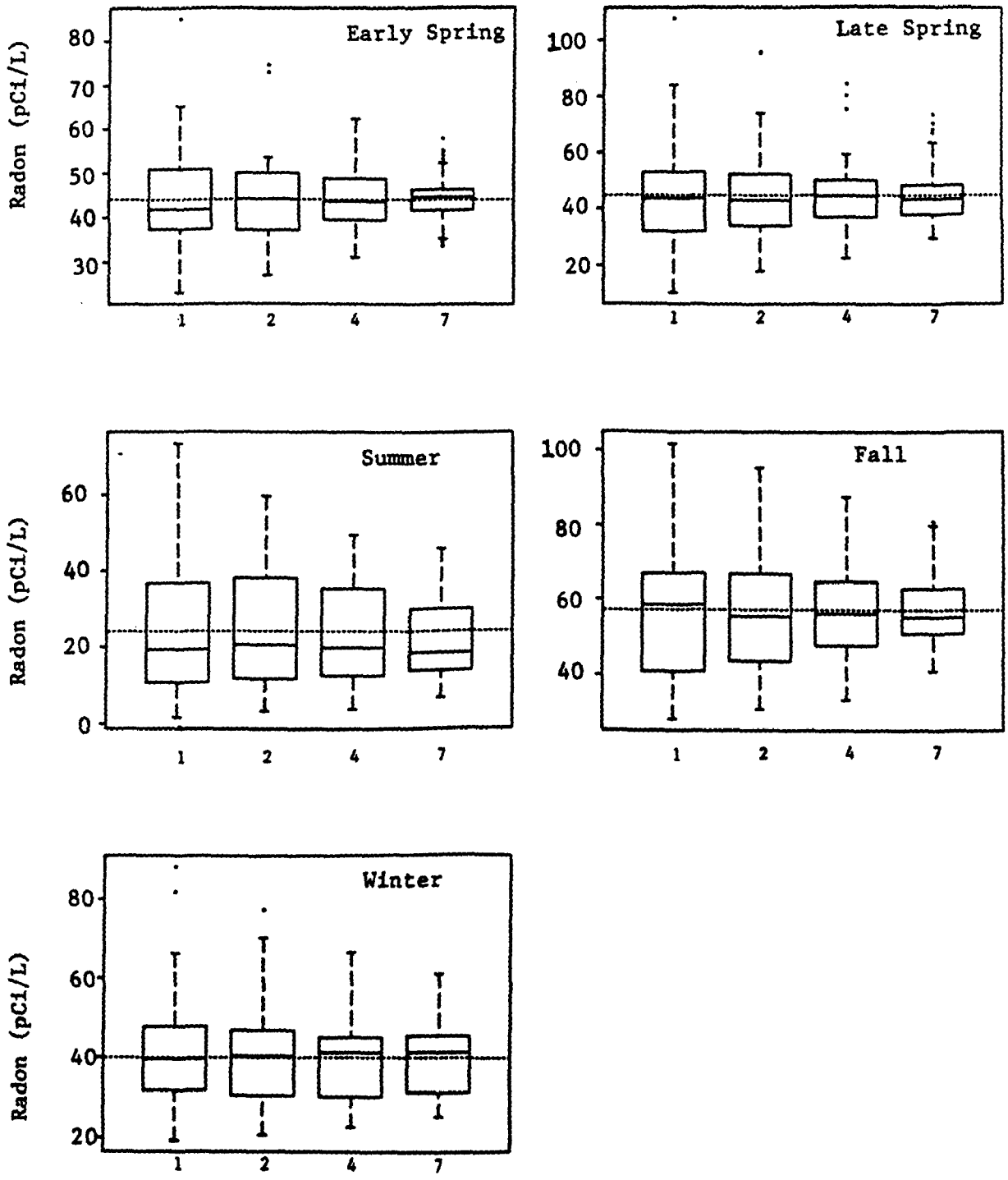

Figure 2 Box and whisker plots of 1-, 2-, 4-, and 7-day averages of basement radon levels for the five different seasons chosen. Dashed lines represent the seasonal average, while the median is shown by the solid line. Outlier data points falling outside the inner fences are also shown.

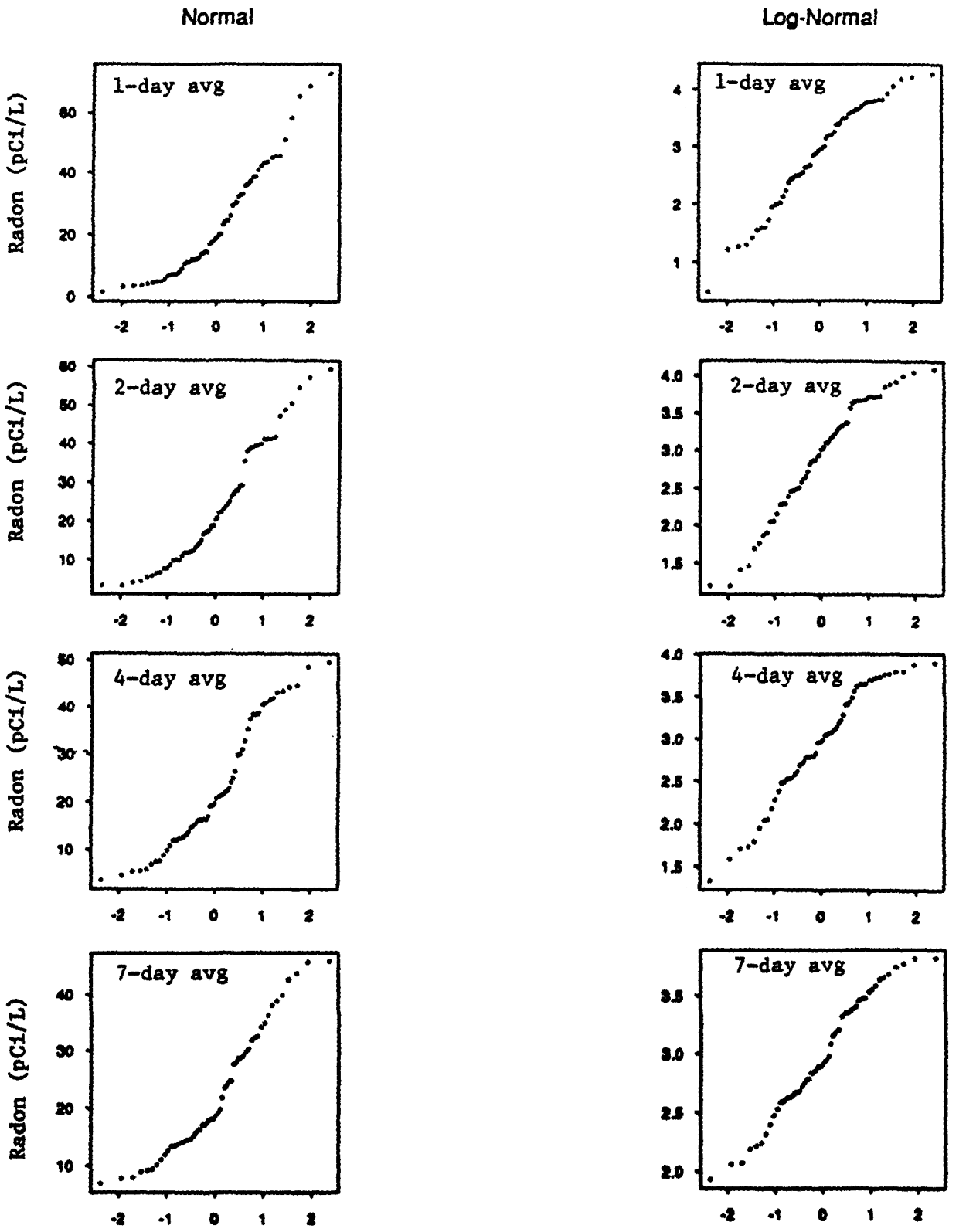

Figure 3a Quantile-quantile plots of basement radon levels for different averaging periods during Summer.

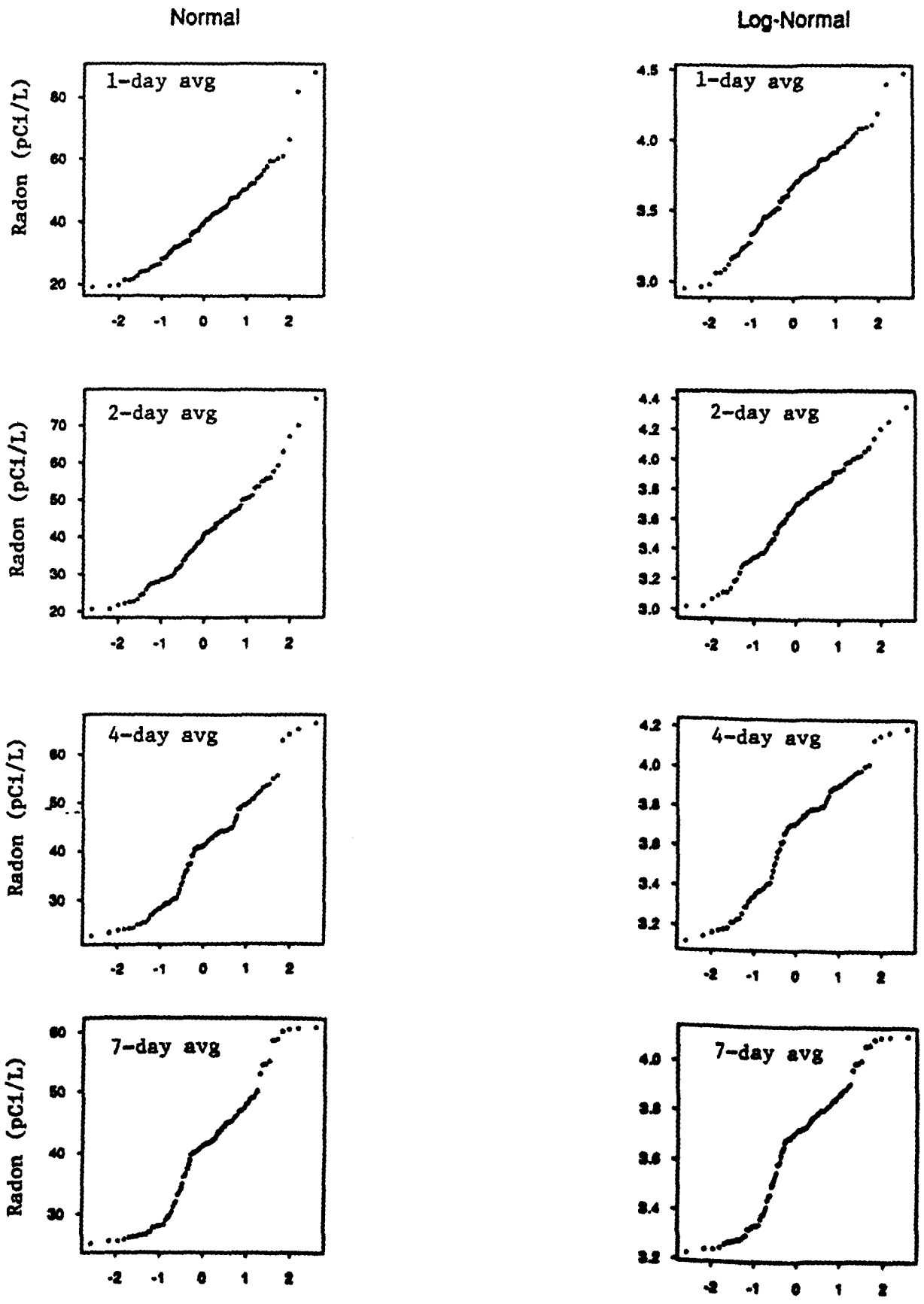

Figure 3b Quantile-quantile plots of basement radon levels for different averaging periods during Winter.

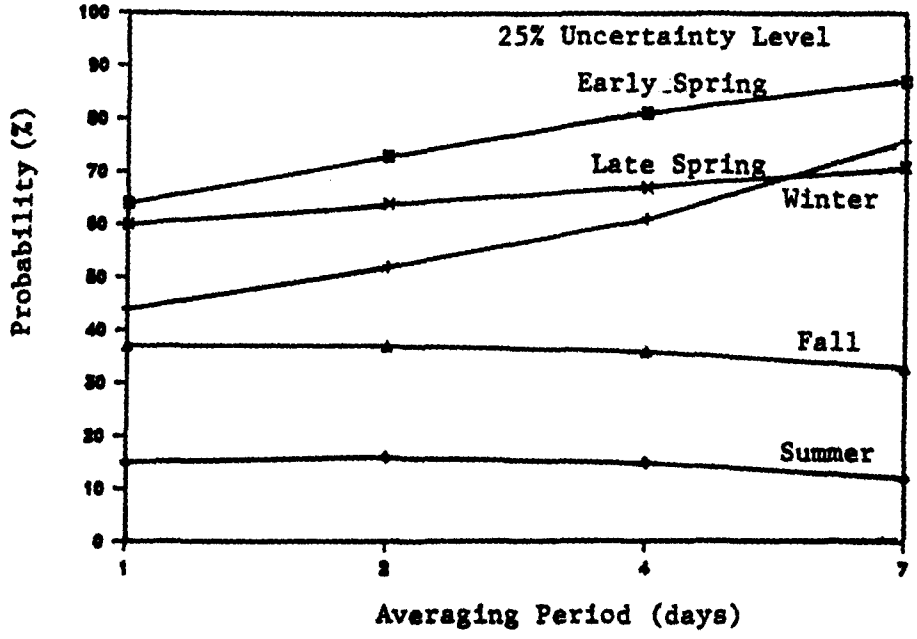

Figure 4 Reliability of single short-term measurements done during different seasons for H22.

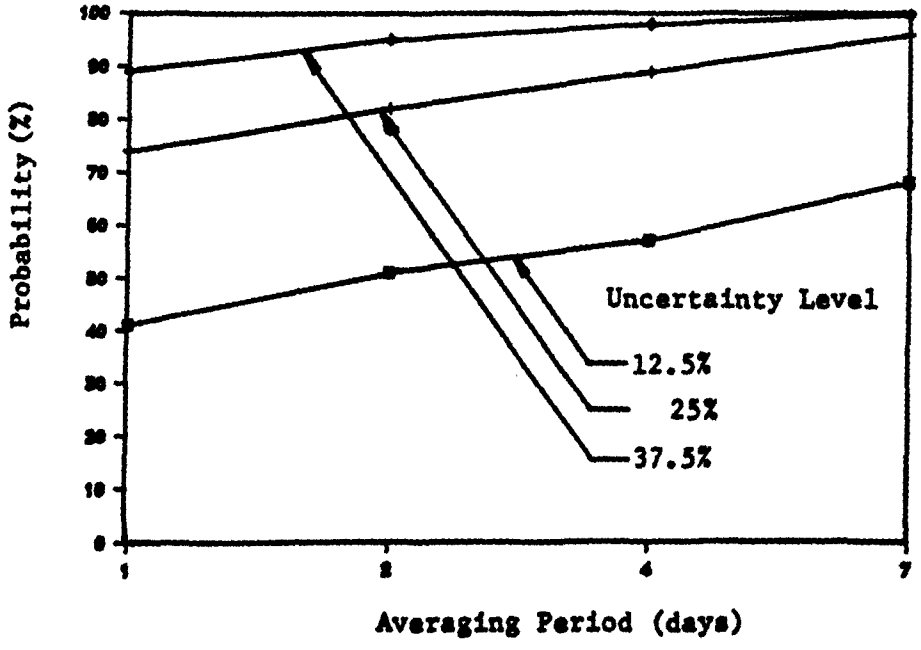

Figure 5 Reliability of a protocol involving four seasonal short-term measurements to predict long-term average of H22.

RESULTS OF SHORT- AND LONG-TERM RADON MEASUREMENTS IN SOIL AND DWELLINGS BY ALPHA TRACK DETECTORS.

L. Tommasino and G.Torri
Laboratorio di Misure, ENEA-DISP,
Via V. Brancati 48, 00144, Rome, ITALY

ABSTRACT

Passive radon monitoring devices, based on damage track detectors, present attractive characteristics not only for long-term (months to years) measurements but also for short-term (days to weeks) exposures in dwellings.

A gas radon permeation-type sampler based on a heat-sealed polyethylene bag has been extensively used in Italy for large scale surveys of both soil radon and long-term measurements in dwellings.

For short-term radon measurements, the radon sampling device is obtained by using a bare cellulose nitrate detector. Any desired sensitivity can be achieved simply by scanning a sufficiently large detector area by the the spark counter.

Several different sets of data have been gathered so far using both the techniques mentioned above, which data are very useful to illustrate the weekly, monthly and yearly variation of the radon concentration respectively in soil and in dwellings.

INTRODUCTION

There are a wide variety of well established techniques for the measurement of radon and its daughters (1). In the following we will describe the radon monitoring by alpha track detection with the specific applications in mind of large scale surveys, i.e. those applications where damage track detectors present unique characteristics. Large scale surveys for the assessment of the radon concentration in dwellings are needed respectively for:

- The assessment of long-term radon exposures (months to years) for determining the population doses.
- The assessment of short-term radon exposures (days to weeks) for screening purpose or for investigating the geographical variation of the radon concentration.

ASSESSMENT OF LONG-TERM EXPOSURES

THE PLASTIC-BAG RADON SAMPLER.

Damage track detectors present unique characteristics for the assessment of long-term radon exposures for their ability to integrate over long periods of time (months to years). Radon monitoring devices based on damage track detectors can be formed respectively by diffusion and permeation samplers (2) .

A novel radon monitor device has been recently developed (3,4) which is a permeation sampler formed by a heat-sealed plastic bag made of polyethylene. Enclosed in the bag are two LR-115 detectors held one cm apart by a plastic frame, the cross-section of which is shown in figure 1. Aluminized polycarbonate degraders are facing the detectors both to optimize the detector responses and to make their surfaces conductive. Of the two LR-115 foils, only one is used for the assessment of the radon concentration while the other is used as the back-up detector.

The back-up detector can be made of any type of detectors, such as chemically and/or electrochemically etched polycarbonate or CR-39 detectors.The polyethylene bag is needed to protect the detector from humidity, dust, thoron and radon daughters, allowing the preferential entry of radon gas(2,3). This bag is very important for long exposure periods both in indoor and outdoor environments and presents the following advantages:

- low cost, simple heat-sealing;
- small size and fast sampling time;
- high permeability to radon with small permeability to water vapor;
- no formation of glass-like or brittle like bag in very cold weather.

SOIL AND INDOOR RADON MEASUREMENTS WITH THE PLASTC BAG SAMPLERS.

The most important contribute to indoor radon is the radon of the soil beneath the house. Extensive investigations of soil radon have been made in Italy in the past few years mainly using the plastic bag sampling device (5,6). For soil radon measurements, detectors have been placed at a depth of 0.6 m below the ground surface. Figure 2 shows how important is the protection of the polyethylene bag under large humidity conditions. In this figure two replicas are shown, which have been obtained by spark counting LR-115 foils (7) irradiated respectively with (right-hand side) and without (left-hand side) protection of the polyethylene bag. The water droplets on the detector surface can be easily seen, which droplets greatly affect the response of the radon sampler.

The survey of soil radon in Italy commenced in November 1982

by establishing hundreds of different monitoring stations mainly in Latium and Campania, which are two regions of Italy with rather large content of uranium in soil. Some of these field stations are still in use after an interruption of two years (1984, 1985). The seasonal variations of soil radon in these stations are shown in figure 3, which reports the mean radon concentration versus the month period for four stations in the year of 1983, 1986, and 1987 (6). The radon concentrations have been normalized to a month arbitrarily chosen (August 1983). It appears clear from this figure that there is a constant trend for the three different years. The radon concentration results always higher in the summer and in the fall. It can be also noted that the total radon concentration relative to the first 9 months of 1987 is 3 times higher than that of 1983. Long term variations of soil radon indicate that indoor radon concentrations may be known with a limited accuracy even when radon measurements are integrated over an entire year (6).

The plastic bag radon sampler is now being used for the italian national survey of indoor radon, which is carried out with the contribute of different regional laboratories. The simplicity, the low cost of both the radon sampler and the track counting automation-system have greatly facilitated the use of this technique by different italian laboratories.

Finally, radon monitoring in a roman residential home has been made continuously since 1987. Figure 4 shows the monthly variation of indoor radon concentration in the children bedroom of this test house. During the cold weather registered on January 1987 (when Rome was covered by almost one meter of snow) the radon concentration was relatively high. In the holiday period of August 1987, the bedroom was kept closed and the radon concentration increased as a result of a decreased air exchange rate. Since then the radon concentration was never so high because of the exceptional good weather, registered in Rome both in 1988 and 1989.

SHORT-TERM RADON MEASUREMENTS

LIMITATIONS OF EXISTING TECHNIQUES FOR SHORT-TERM MEASUREMENTS.

Measurements of short-term exposures (from a few days to one week) are particularly useful for screening surveys to identify houses with high radon concentrations and to investigate geographical variations. Unfortunately all the above radon gas samplers are not sufficiently sensitive for short exposure-periods. For these applications the Diffusion Barrier Charcoal Adsorption Collector-DBCA is used. Cohen (8) has reported an exercise based on the assumption of the validity of the Poisson statistics in order to compare uncertainties (one standard

deviation) between SSNTD samplers (used for three months exposures) and the diffusion barrier charcoal adsorption method - DBCA (one week exposure). This comparison results in the following:

Bq/m3	DBCA (one week)	Microscope (three months)
18.5	46%	89%

These conclusions are misleading since they are based on less than 10 mm^2 of microscope counting area. On the other hand, areas of hundreds of cm^2 can be easily scanned in a fraction of a second using the spark counter (7). Since it was suggested that there was no alternative to the charcoal adsorption technique, this latter technique has been widely used for screening surveys in spite of the following limitations (9):

- Response with undue weight to the final day or two of the exposure;
- Batch to batch variations;
- Response highly dependent on temperature and humidity.

In the following it will be illustrated how simple it is to measure short-term radon exposures in dwellings by using the spark counter for the scanning of large detector-areas of cellulose nitrate foils.

LOWER LEVEL OF DETECTION-LLD

Assuming valid the Poisson statistics, the Lower Level of Detection - LLD can be related to the area of detector counted, S, as in the following (10):

$$LLD = 4.6 \frac{\sqrt{\rho S}}{\varepsilon S} \qquad (1)$$

where ρ is the background track density and ϵ is the track registration efficiency per unit exposure.

According to equation (1), if the detector area is increased by a factor of one hundred, the LLD decreases by a factor of ten. Since detector areas up to hundreds of cm^2 can be easily counted with the spark replica counter (7), there should be no problem of measuring short-term radon exposures by damage track detectors.

THE ENVELOPE-TYPE RADON SAMPLER

To be able to use large detector areas, a radon monitor based on bare cellulose nitrate foils appears the most convenient (2,11). This detector presents a response directly proportional to the radon gas concentration with little dependence on the plate-out and the equilibrium factor (12,14).

To exploit these attractive characteristics a new radon monitor has been developed using a large cellulose nitrate film (LR-115 Type II with an area of 9cmx12cm) enclosed in an envelope (2,11). When the envelope is closed the LR-115 detector does not register alpha-particle tracks and the radon monitor is in the off-state. To turn it on, the envelope must be opened as shown in Figure 5. Half of the strippable LR-115 foil is protected by a plastic cover thick enough to stop all alpha particles from radon and its daughters. This part of the LR-115 foil represents the "blank" detector, while the remaining bare part is used for the radon gas measurement. After exposure, the entire detector is etched with 2.5 N NaOH solution at 60 °C for 110 minutes. Once etched and washed, the thin cellulose nitrate film is stripped off from the backing and both the "blank" and the bare detector surfaces are counted with a 43 cm^2 electrode at 500 volts after a pre-sparking at 900 volts. This counting procedure takes less than one second and can be repeated up to hundreds of times with a reproducibility better than 2 % .

In spite of the very different etching and counting procedures, the registration efficiency, ϵ, has resulted in a value similar to that evaluated by other investigators (13,15) and precisely:

$$\epsilon = 2.0 \text{ (Tracks . m3.h)/(cm}^2\text{. kBq)}$$

Comparison of the advantages and the disadvantages of the SSNTD and the DBCA methods for indoor radon measurements have been reported (8) under the assumption of the validity of Poisson statistics. When similar comparison is made between the DCBA and the spark counted envelope detector, the statistical uncertainties (one standard deviation) of measurements for one week exposure in indoor air result as in the following:

Bq/m3	DBCA	Spark Counter
18.5	46 %	12 %

Incidentally the small uncertainties predicted on the basis of the Poisson statistics have been confirmed by preliminary empirical data.

To conclude, the envelope-type radon sampler presents the following advantages:

- Possibility of scanning large detector areas in a fraction of second achieving very low detection limits.

- Possibility to turn off and on the detector simply by closing and opening an envelope.
- Simplicity of delivering the detector by mail.
- High sensitivity and accuracy for short term exposures.
- Unique possibility to have a "blank" detector.
- No diffusion time required for the radon to enter the sampler volume.

Figure 6 shows the weekly variations of the radon concentrations in the bedroom of the roman test house in the spring 1988. Variations greater than a factor of two are measured, which variations can not be monitored with the plastic bag radon sampler.

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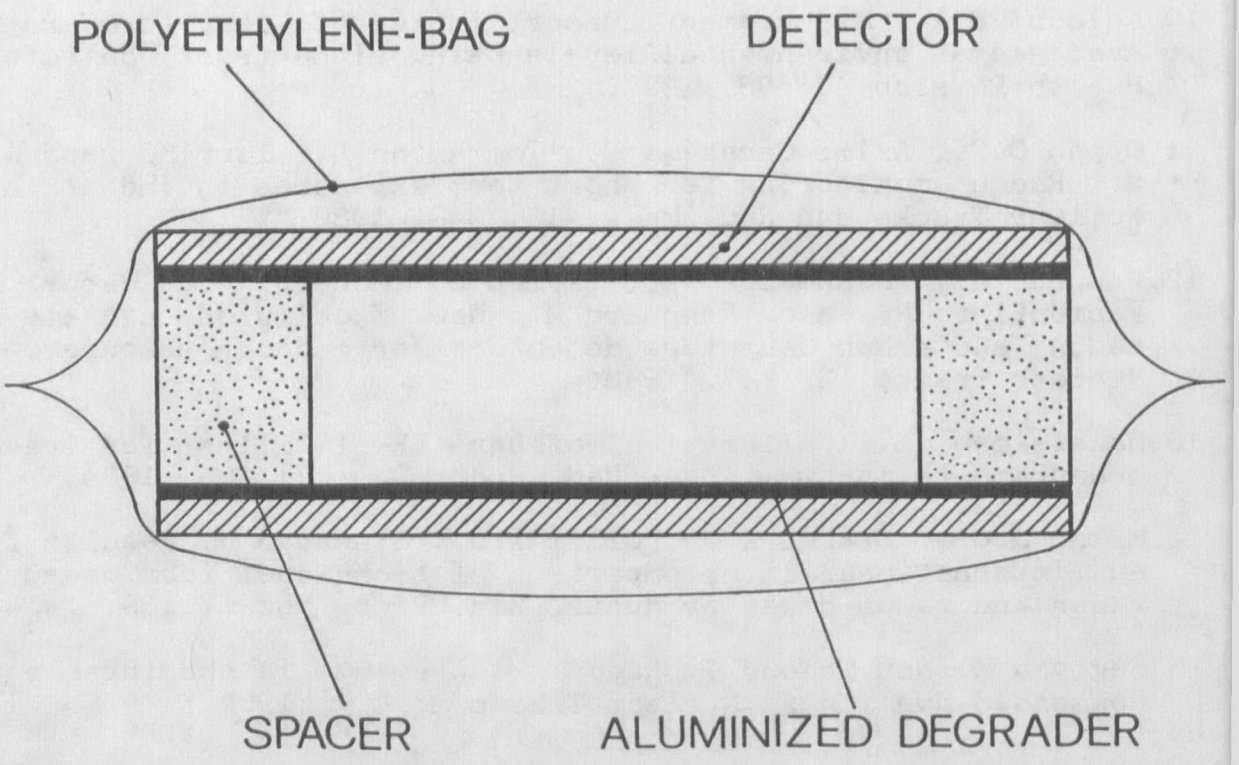


FIG. 1 THE PLASTIC-BAG RADON-GAS SAMPLER

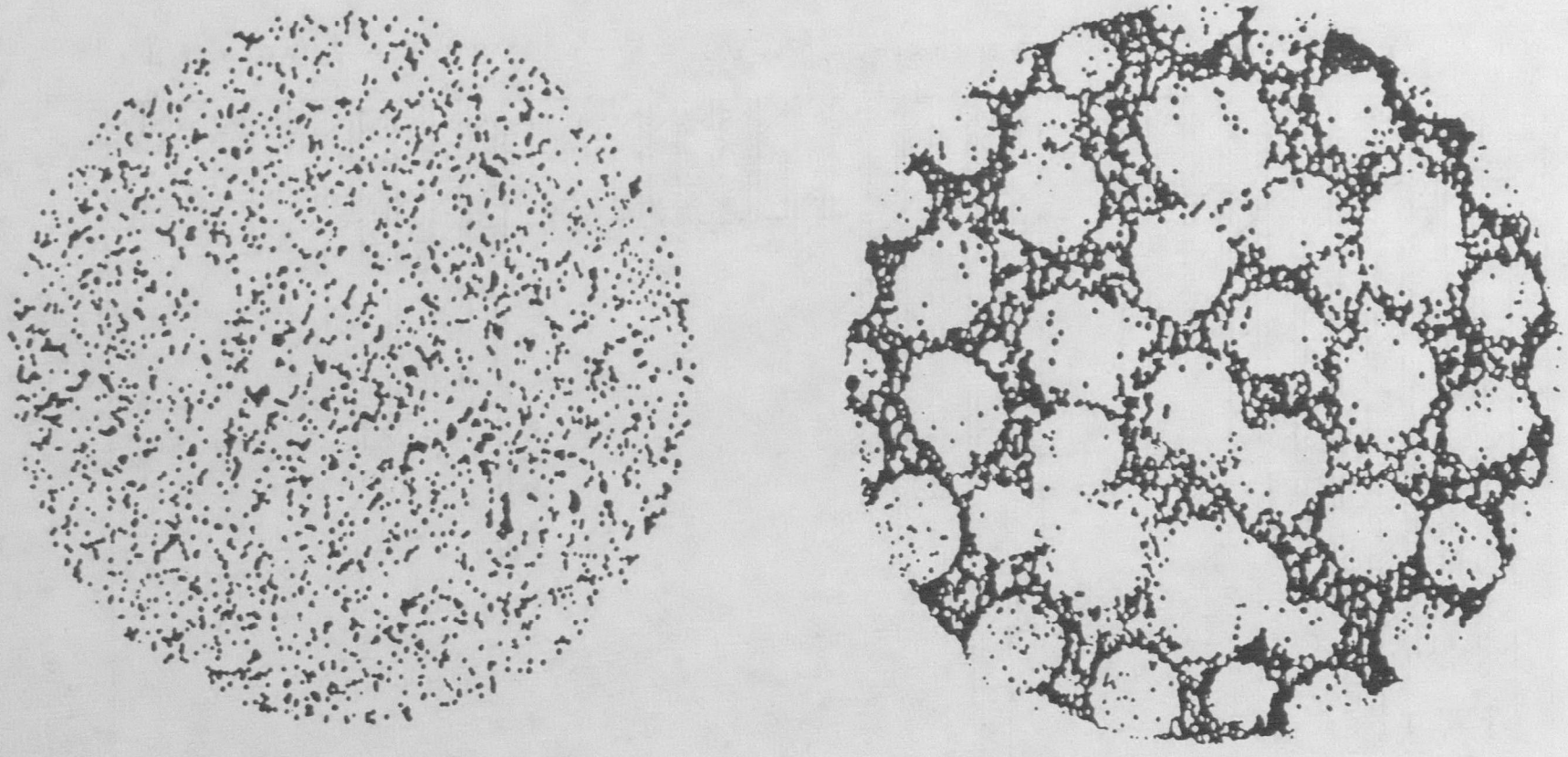


FIG. 2 SPARK COUNTER REPLICA OF LR-115 DETECTORS WITH AND WITHOUT PLASTIC BAG.

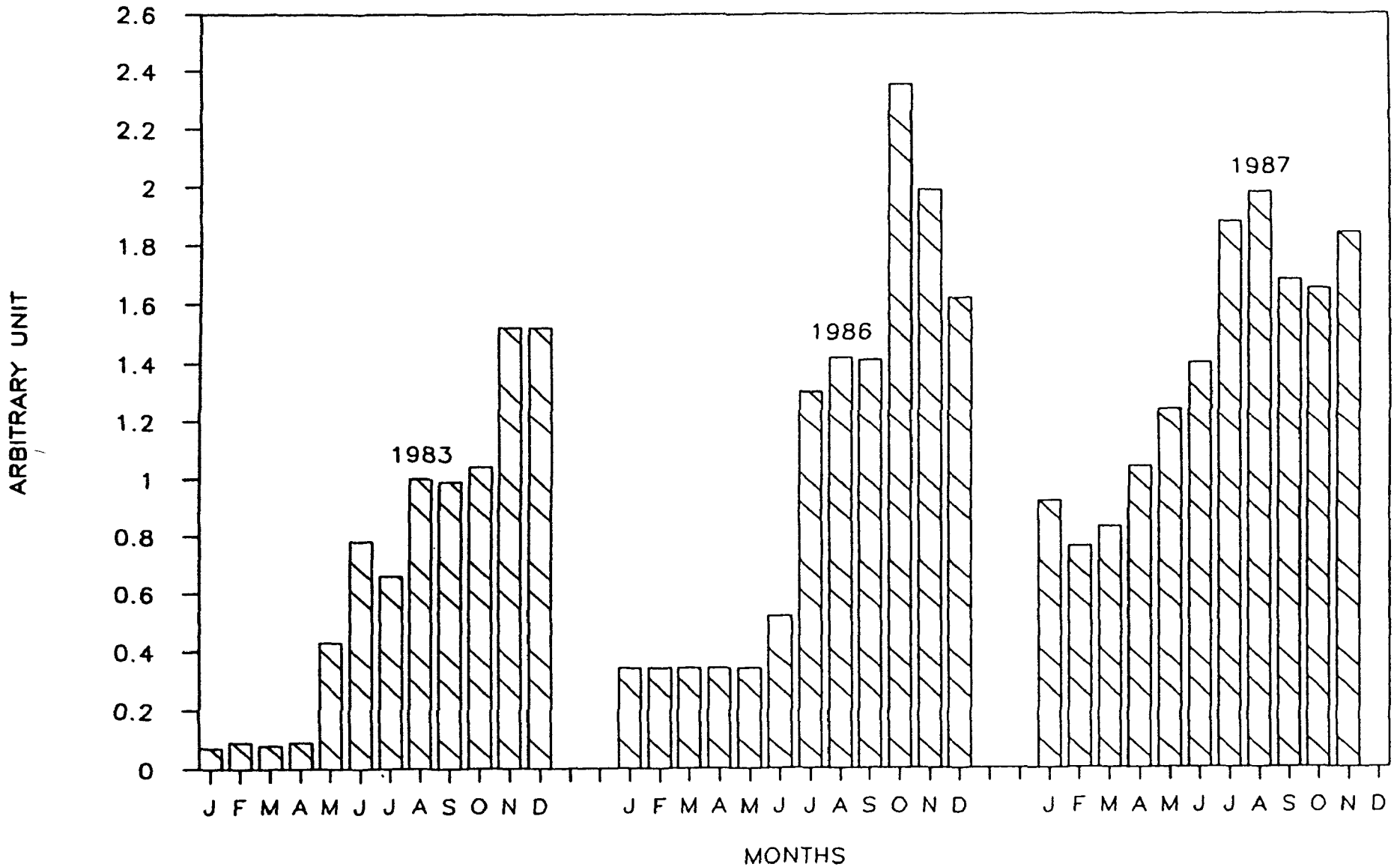


FIG. 3 SEASONAL VARIATIONS OF SOIL RADON

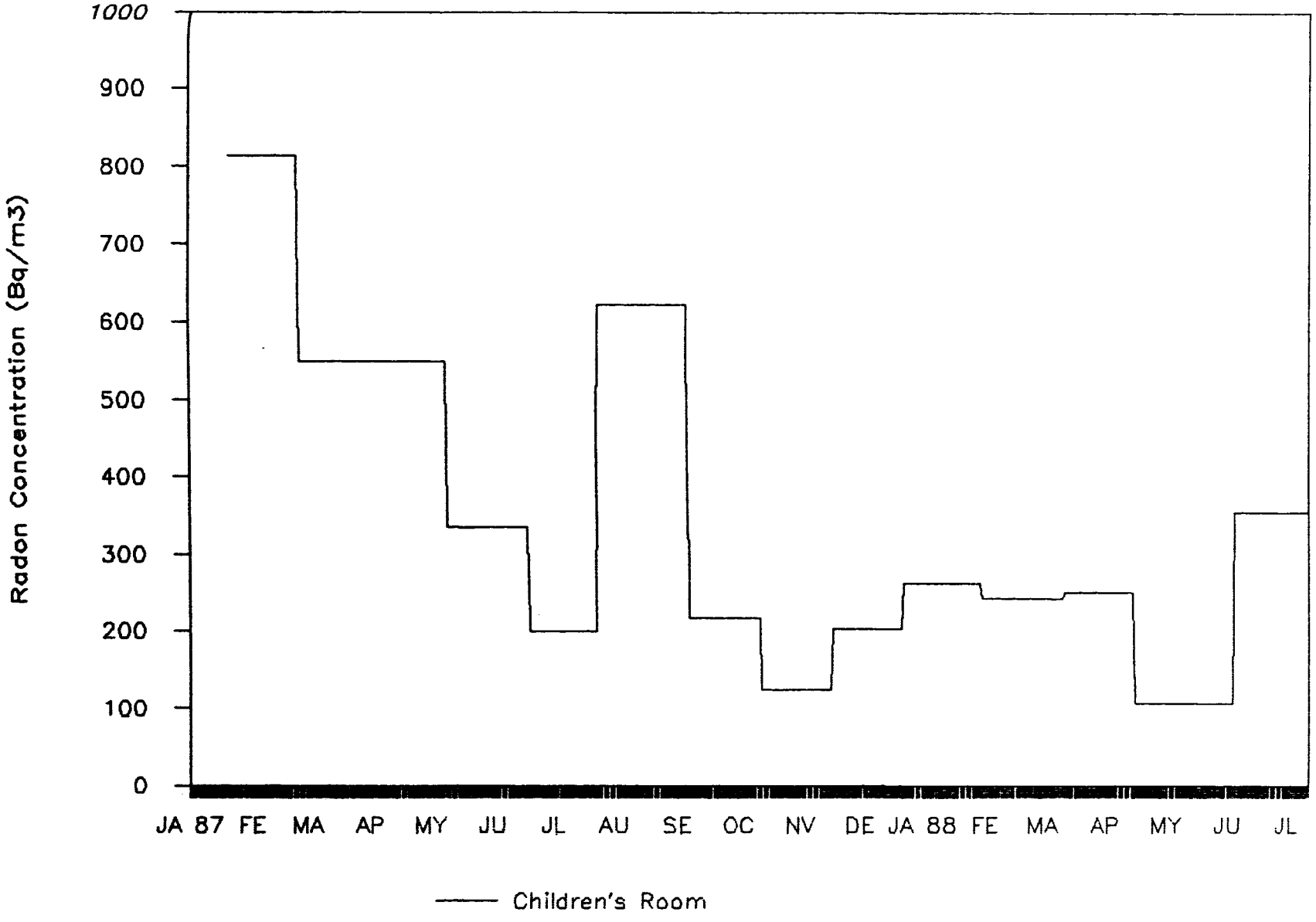
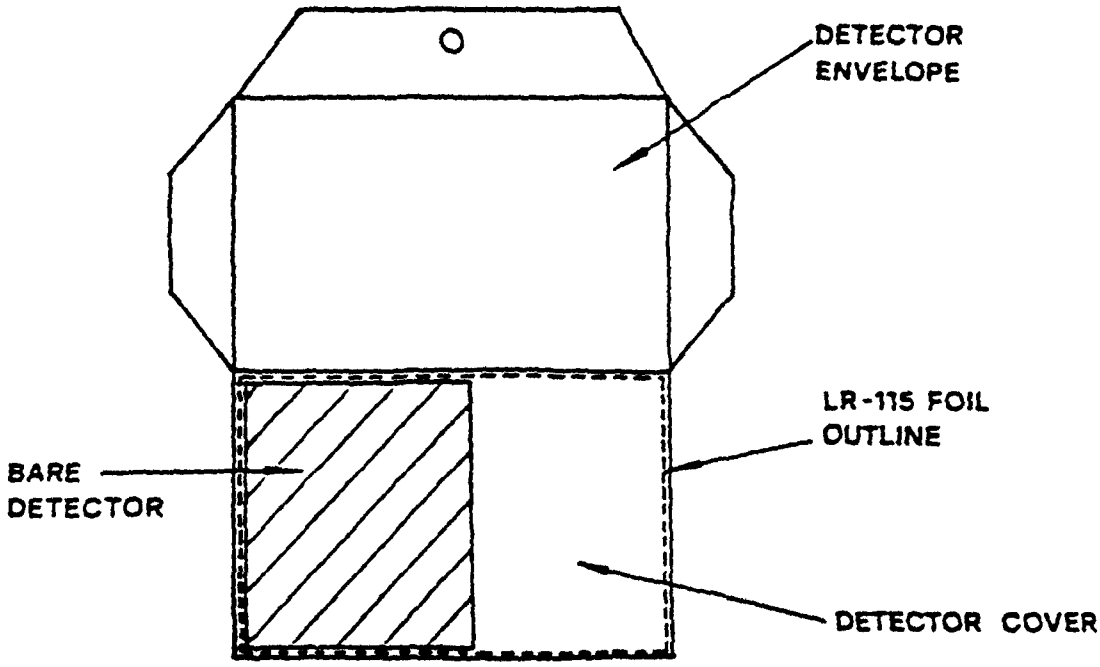


FIG. 4 MONTHLY VARIATION OF THE RADON CONCENTRATION IN THE TEST-HOUSE.



ENVELOPE-TYPE RADON MONITOR

FIG. 5 ENVELOPE-TYPE RADON GAS DETECTOR

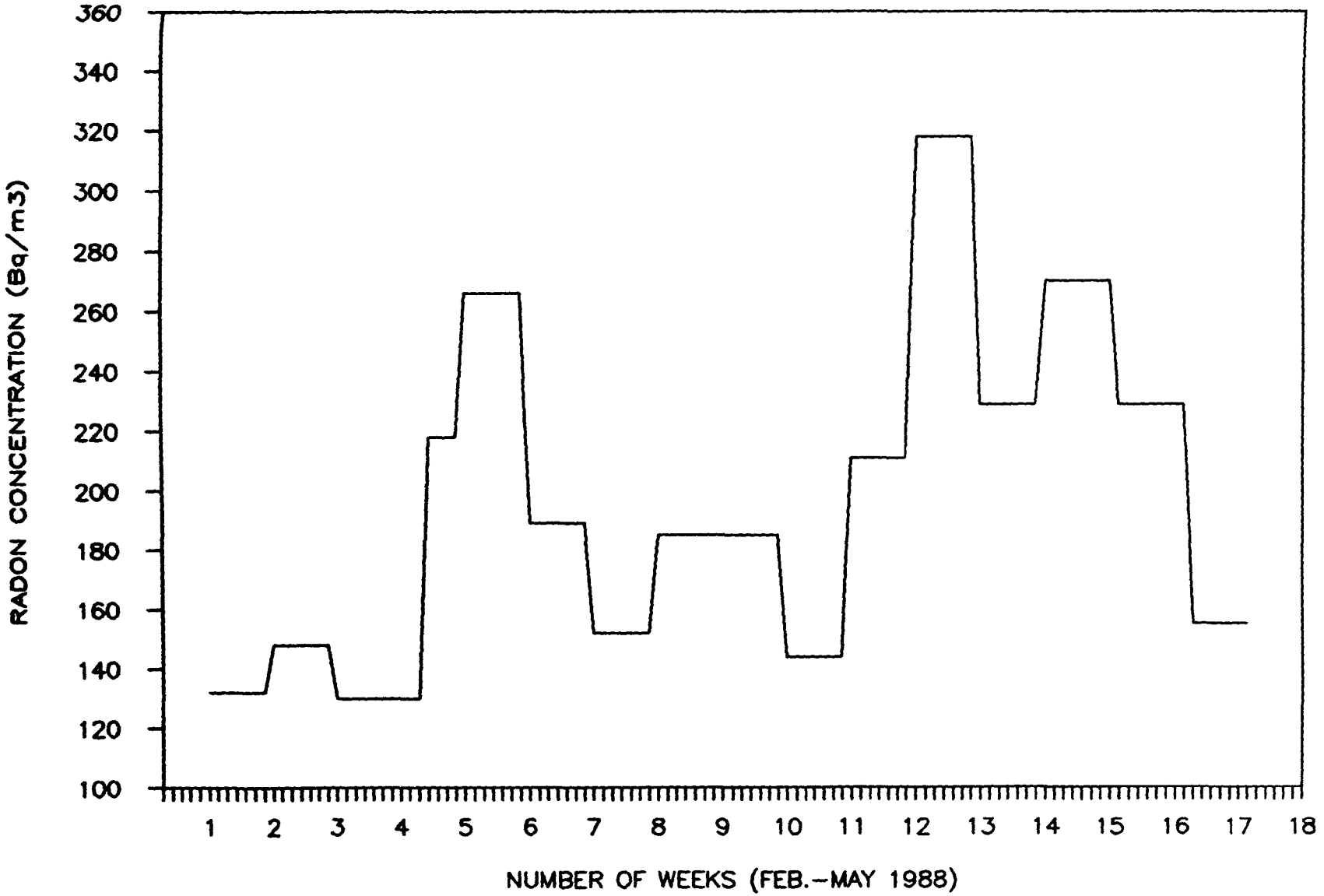


FIG. 6 WEEKLY VARIATION OF THE RADON CONCENTRATION IN THE TEST HOUSE.

A STATISTICAL ANALYSIS: PREDICTING ANNUAL ^{222}Rn CONCENTRATIONS FROM 2-DAY SCREENING TESTS

S.B. White, C.A. Clayton and B.V. Alexander
Research Triangle Institute
Research Triangle Park, N.C. 27709

M.A. Clifford
U.S. Environmental Protection Agency
Office of Radiation Programs, Washington, D.C.

ABSTRACT

Radon screening tests are short-term tests used to determine whether additional testing for longer duration (usually one year) is needed to more accurately characterize health risks to Rn exposure. The extent to which these tests are useful in reaching appropriate decisions is governed by the degree to which short- and long-term measurements are related. A subsample of participants in the EPA/State Indoor Radon Surveys for 1988 provided Rn measurements from 2-day charcoal canisters located in the lowest livable level and from 1-year alpha track detectors placed on each livable level. Data from 383 houses located in six states and representing a broad range of geologic conditions, climates, and housing types were used to derive the relationship between 2-day measurements and 1-year measurements averaged over all levels to represent the household annual concentration. For given short-term concentrations, the ability to predict annual average concentrations for an individual house and the likelihood of errors in misclassification are examined.

The relationship between short-term screening measurements and long-term annual measurements is an important parameter in developing program policy and public information materials. This statistical analysis is one of a series of studies being undertaken by the EPA to look at this relationship.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Short-term screening tests for radon are used to determine if additional testing (usually one year duration) is needed to more accurately characterize health risks to Rn exposure. The extent to which screening tests can properly identify houses needing further testing is governed by the degree to which short- and long-term measurements are related. The relationship between short- and long-term radon measurements has not been studied extensively. In a recent review of the published literature, Ronca-Battista (1) found only nine studies in which this issue was addressed and six of these had sample sizes less than 100 houses. This statistical analysis is a part of a series of studies currently being undertaken by the U.S. Environmental Protection Agency (EPA) to better understand this relationship.

A subsample of houses in the EPA/State Indoor Radon Surveys for 1988 was tested with a 2-day charcoal canister placed in the lowest livable level and with 1-year alpha track detectors (ATDs) placed on each livable level. A total of 383 houses located in six states provided both short- and long-term radon measurements. Relationships between 2-day measurements and 1-year measurements were derived for four different housing types and for all houses combined. The ability to predict an annual concentration for an individual house is examined. Also, the likelihood of making incorrect decisions (i.e., false positive and false negative errors) using screening measurements is investigated.

OBJECTIVES

The purposes of this study were (1) to examine the overall relationship between short-term (i.e., 2-day) measurements and long-term (i.e., 1-year) measurements of indoor radon, (2) to determine if a short-term measurement can be effectively used to predict a long-term measurement for an individual house, and (3) to assess false positive and false negative error rates associated with a short-term screening test.

METHODOLOGY

Two indoor radon measurements (X, Y) were obtained on each house participating in this study. X refers to a 2-day charcoal canister reading taken on the lowest livable level according to the EPA's recommended protocol. Y refers to the house annual concentration obtained by averaging 1-year ATD readings taken on each livable level (two ATDs were used in one-story nonbasement houses). There are several other methods which could be used for characterizing the annual concentration. One would be to use the first floor ATD measurement. Another would be to use a weighted average of the ATD measurements from each level, where the weights reflect the proportion of time spent on each level. This paper has been limited to the average of the measurements taken on each livable level.

The houses selected for this study were of four types - one-story with basement, two-story with basement, one-story without basement, and two-story without basement. A relationship between X and Y was derived for each type of house. This was done in anticipation that the relationships may differ substantially from one house type to another and, if true, separate prediction equations would be required. In all analyses, X is the independent or predictor variable and Y is the dependent variable.

A scatter plot of the data indicated that Y was linearly related to X and that the variation in Y tended to increase as X increased. The relationship between X and Y was derived using a model which

reflects these visual observations in the data. A specification of the model is given below.

The results in this paper employ a mathematical model that assumes that long-term measurements of radon (1) are linearly related to short-term measurements and (2) have variances that are proportional to their expected values. That is,

$$Y_i = (\alpha + \beta X_i) + \sigma Z_i(\alpha + \beta X_i)^{1/2} \tag{1}$$

where

Y_i = ATD measurement on the ith house, (average over all floors),

X_i = canister measurement on the ith house (lowest livable level),

α, β, σ = parameters to be estimated, and

Z_i = random error for ith house, assumed to be normally distributed with mean 0 and variance 1.

In order to convert (1) to a model having a homogeneous error structure, we divide by $(\alpha + \beta X_i)^{1/2}$ and substitute $\sqrt{Y_i}$ for $(\alpha + \beta X_i)^{1/2}$ on the left hand side of (1):

$$\sqrt{Y_i} = (\alpha + \beta X_i)^{1/2} + \sigma Z_i. \tag{2}$$

The parameters in (2) were estimated using nonlinear least squares. The prediction equation $\widehat{\sqrt{Y_i}} = (\hat{\alpha} + \hat{\beta} X)^{1/2}$ was squared to obtain predictions of the long-term concentrations for given short-term measurements.

Similarly, endpoints of the 95% confidence interval for the expected value of $\widehat{\sqrt{Y}}$ were squared to obtain a corresponding interval estimate for the long-term concentration.

DATA

Data used in this study were obtained from a subsample of 531 homes surveyed in the EPA/State Residential Radon Surveys in six states (Arizona, Massachusetts, Minnesota, Missouri, North Dakota, and Tennessee). In December 1987, the states began placing one charcoal canister and from two to four alpha track detectors in survey eligible homes. The homeowners were instructed to expose the charcoal canister for a two-day period on the lowest livable level of their home during closed-house, winter conditions. At the same time, the homeowners were supplied with enough alpha track detectors to place at least one detector on each livable level of their home. Homes with only one livable level were given two alpha track detectors for exposure in different rooms on the one level. Homeowners were instructed to expose the alpha track devices for a one-year period and were contacted by the state at the end of the exposure period to remind them to return the devices.

For purposes of this investigation, the data were restricted to four types of homes: one-story with basement, two-story with basement, one-story without basement, and two-story without basement. After examining the data, three homes were excluded as outliers (one each in Missouri, North Dakota, and

Tennessee). In addition, all homes with a canister reading less than 1 pCi/L or greater than 35 pCi/L were excluded from the analysis so that the prediction equations would not be influenced by values in this range. After applying these restrictions, 383 homes were included in the analysis. Table 1 gives the number of homes tested and the actual number included in the analysis from each of the six states. It should be noted that 136 homes were excluded due to canister measurements less than 1 pCi/L, whereas only 9 homes were excluded due to canister readings greater than 35 pCi/L.

Table 2 summarizes the placement of the charcoal canister and the alpha track detectors in the four types of homes. For each house, a new variable was constructed from the mean of the two or three alpha track measurements made on the appropriate levels of the house. This mean value represents the annual household concentration, and it is this variable which was used as the long-term measurement in this study.

RESULTS

The results of fitting the model previously described to each of the four types of houses are shown in Table 3. For each house type, Table 3 gives the prediction equation, the correlation between X and Y, and the standard deviation, $\hat{\sigma}$, from the fitted model.

Note the prediction equations in Table 3 are, for all practical purposes, the same for the four house types. The estimated coefficients of X ranged from 0.5 to 0.6. Also, the correlations between X and Y are similar for the four house types - ranging from 0.79 to 0.82. Finally, the standard deviations of the fitted models for the four house types do not vary enough to warrant separate prediction equations for the four house types. Thus, it was decided to ignore house type and to use only one relationship for all houses.

The results of fitting the model described earlier to all 383 houses is shown at the bottom of Table 3. The prediction equation is

$$\hat{Y} = 0.42 + 0.56X$$

where Y is the expected (or predicted) value of the annual concentration in a house that has a screening measurement of X. For example, for a house that had a charcoal canister measurement of, say, 3.5 pCi/L, the predicted house annual average concentration from 1-year ATDs placed on each level would be 2.4 pCi/L. (= 0.42 + 0.56(3.5)).

A scatter plot of the actual test results from all 383 houses is given in Figure 1. Note that most of the houses reflect a canister measurement under 6 pCi/L and a long-term measurement of the house annual concentration under 4 pCi/L. Based on other survey results and research findings this was expected. Note also that as the short-term measurements get larger the long-term measurements show greater dispersion. This increase in variability in long-term measurements is taken into account by the model used in data analysis.

Superimposed on the scatter plot of Figure 1 are three lines. The center line is, as noted, the prediction equation. The other two lines (designated as UCL and LCL) represent the estimated upper and lower 95 percent confidence limits on the predicted value for an <u>individual</u> house. These should not be confused with 95 percent confidence limits on the prediction equation which are, in this case, very close to the center line (within 1 pCi/L of the prediction equation for screening measurements less than 10 pCi/L) because of the large sample size. The interpretation of the confidence limits in Figure 1 is as follows. If a 2-day charcoal canister reading is X for a given house, there is a 95 percent chance that the true house annual average concentration would be covered by the interval falling between the upper and lower lines corresponding to X. For instance, if X = 14, we can be 95 percent confident that the interval (3.7 - 14.4)

will cover the true long-term concentration.

As an aid in utilizing the information shown in Figure 1, Table 4 gives the predicted concentration for an individual house and associated confidence limits corresponding to various short-term screening measurements ranging from 1 to 35 pCi/L.

Screening tests for radon should be used only for determining if additional testing (of longer duration) is needed. EPA currently recommends additional testing if the screening test measurement exceeds 4 pCi/L. In this case, a perfect screening test would correctly classify a house as to whether its annual concentration would exceed 4 pCi/L. Although there is no perfect test, one can, however, assess the performance of a screening test by characterizing the probability or likelihood of an incorrect decision. One of two incorrect decisions can be made on the basis of a screening measurement--if a screening measurement is ≤ 4 pCi/L, one may incorrectly conclude that the house annual concentration is ≤ 4 (false negative); if a screening measurement exceeds 4 pCi/L, one may incorrectly conclude that the house annual concentration is also greater than 4 pCi/L (false positive).

The probability that the house annual concentration will exceed 4 pCi/L, given a specified screening measurement, X, is given by

$$P\left[Z < \frac{(a+bX)^{1/2} - 2}{\hat{\sigma}} \mid X\right]$$

where Z is a standard normal deviate, a and b are the estimated model parameters, and $\hat{\sigma}$ is the standard deviation from the fitted model (bottom line in Table 3). This probability was calculated for screening measurements ranging from 1 to 16 pCi/L; the results are shown in Figure 2. The regions of false positive and false negative errors are noted and the probability of an error associated with a given screening measurement can be determined directly from the plotted curve. For instance, for a screening measurement just below 4 pCi/L the probability of a false negative error is about 0.2. The probability decreases to about 0.05 for screening measurements near 2 pCi/L. On the other hand, for a screening measurement just over 4 pCi/L, the probability of a false positive error is almost 0.8 (one minus the point on the curve at 4 pCi/L). Although the probability of a false positive decreases as screening measurements increase, it is still rather high (about 0.32) for a screening measurement of 8 pCi/L. A high false positive error rate implies that a large number of houses may be needlessly subjected to further testing or corrective action.

False positive and false negative error rates can be changed by changing the screening test criterion for making additional measurements. If the current criterion is changed from 4 to 6 pCi/L, the false negative rate increases and the false positive rate decreases. The magnitude of the errors under such a change can be determined from Figure 2 by shifting the vertical line two units to the right (i.e., centered over 6 pCi/L). Under this rule change, the probability of a false negative error would be about 0.45 for screening measurements just under 6 pCi/L and the probability of a false positive error would be about 0.55 for screening measurements just over 6 pCi/L.

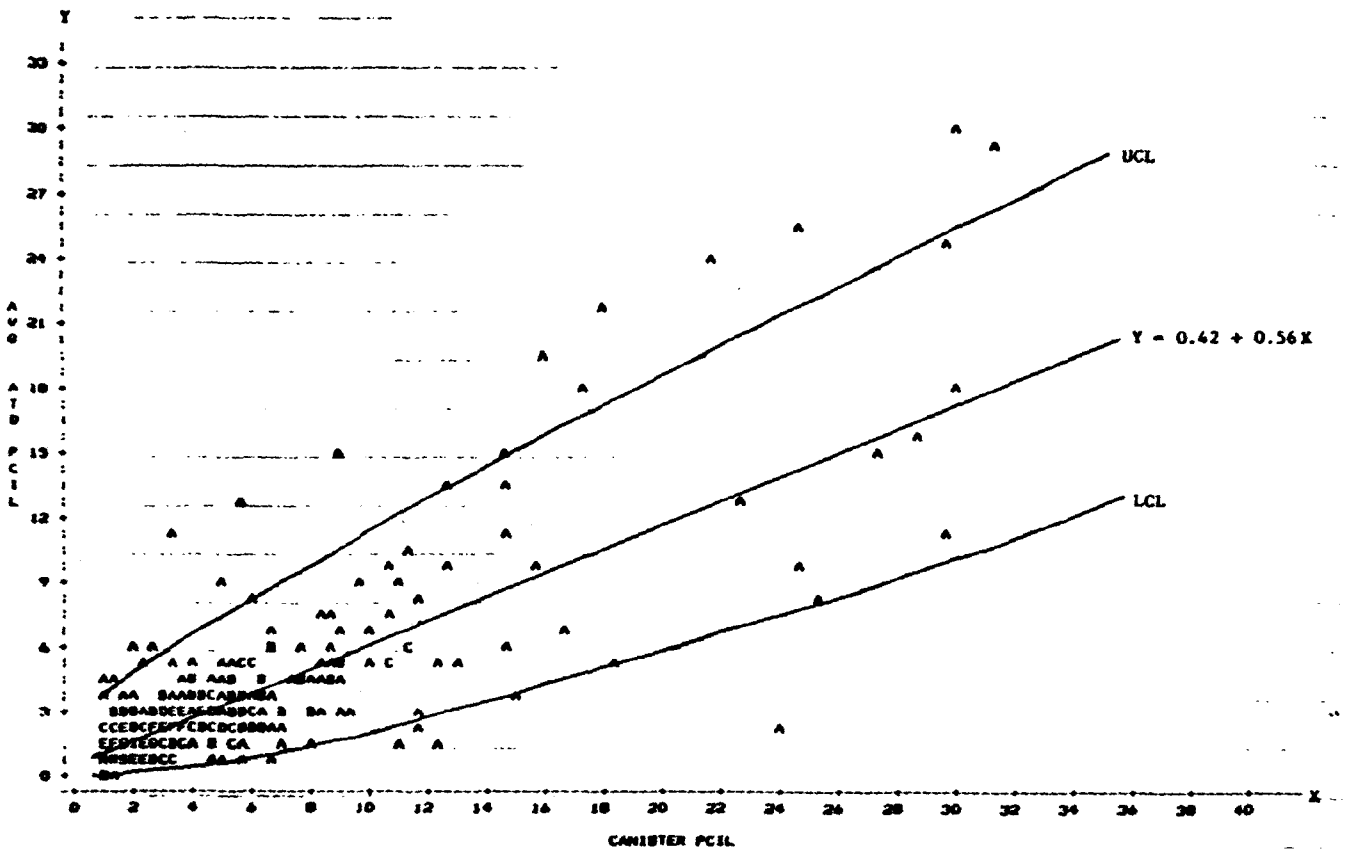
The numbers of false positive and false negative errors that actually occur depend on the distribution of concentrations in the houses being tested. If, for example, most houses have screening measurements under 4 pCi/L then, clearly, the number of false negative errors will be very small. The number of false positive and false negative errors found in the houses included in this study are shown in Table 5. The false negative error rate was 3% (11 out of 354) as contrasted to a false positive error rate of 48% (84 out of 174).

CONCLUSIONS

Two-day charcoal canisters and 1-year alpha track detectors were used to measure indoor radon in 383 houses located over a six state area. Results from this one-year study show there is a strong positive relationship between short-term and long-term measurements - on average, the annual concentration is 56 percent of the screening measurement plus 0.42 pCi/L. The results also show that annual concentration varies widely among houses having the same screening measurement. Consequently, a 2-day screening measurement may not be a very reliable measure of the annual concentration for an individual house. For instance, for a house having a screening measurement of 6 pCi/L, we conclude (with 95% confidence) that the interval (1.0 - 8.2) will cover the true house annual concentration. Finally, if a screening test uses 4 pCi/L as the critical value for determining the need for additional testing, the findings of this study show the test procedure would have a relatively small false negative error rate (about 12 percent for screening measurements near 3 pCi/L) and a relatively high false positive error rate (about 67 percent for screening measurements near 5 pCi/L). These error rates can, however, be changed by changing the screening test criterion for determining when additional testing is needed.

REFERENCE

11] Ronca-Battista, M. Radon in U.S. Homes: A Summary of the Available Literature on Annual and Screening Radon Concentrations, Season Variations, and Differences Between Floors. Unpublished report prepared for U.S. Environmental Protection Agency, Washington, D.C., September 1989.



LEGEND: A = 1 OBS, B = 2 OBS, etc.

Figure 1. Relationship between a 2-day charcoal canister measurement and a 1-year ATD measurement.

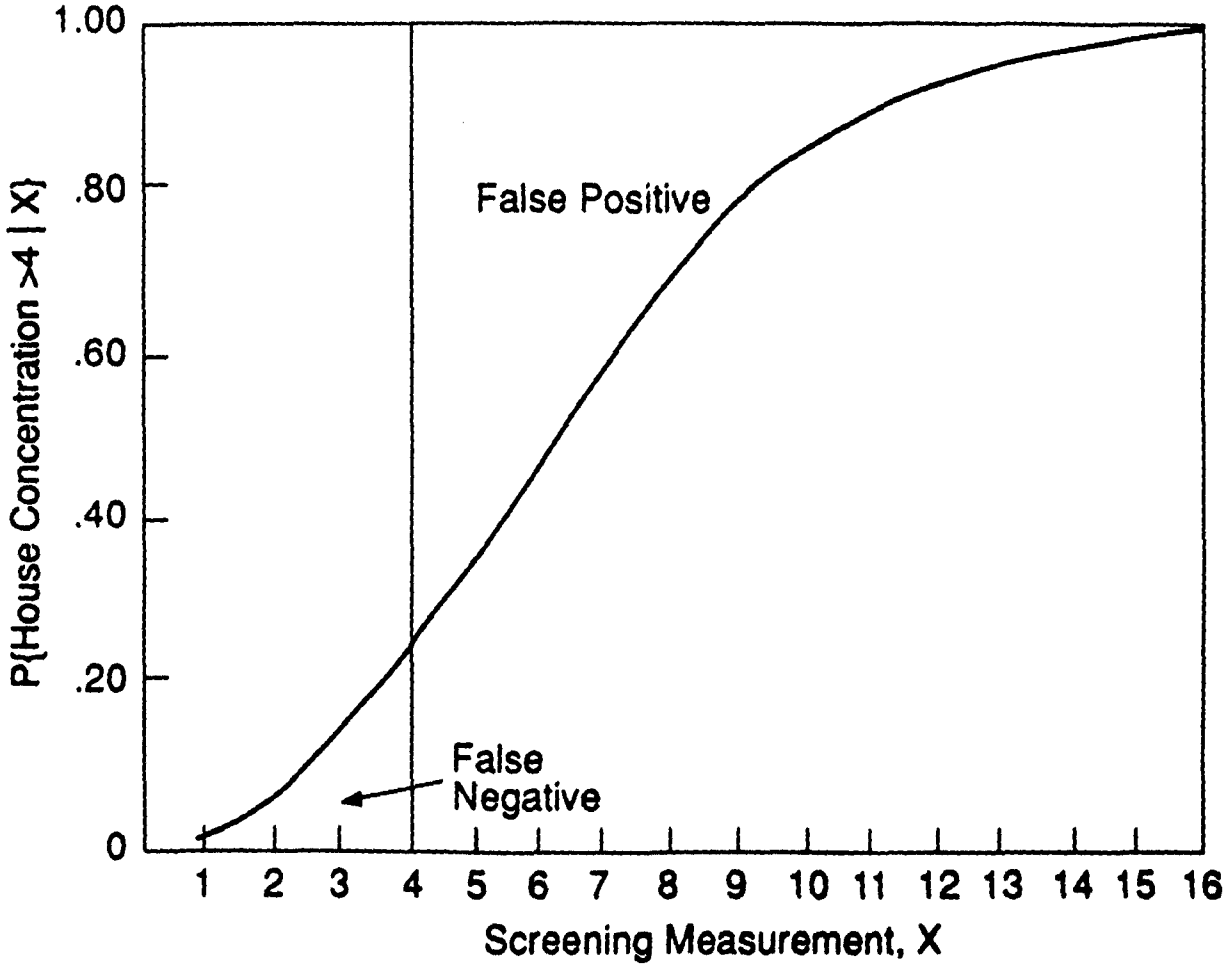


Figure 2. Probability that house concentration exceeds 4 pCi/L as a function of screening measurement.

TABLE 1. NUMBER OF HOUSES TESTED BY STATE

State	Number of Houses Tested	Number of Houses Used in Analysis
Arizona	102	46
Massachusetts	68	51
Minnesota	58	52
Missouri	118	69
North Dakota	119	109
Tennessee	66	56
Total	531	383

TABLE 2. PLACEMENT OF CANISTER AND ALPHA TRACK DEVICES

	Number of Houses	Placement of Canister	Placement and Number of ATDs: Basement	1st Floor	2nd Floor
One-story with Basement	182	Basement	1	1	
Two-story with Basement	71	Basement	1	1	1
One-story without Basement	104	1st Floor		2	
Two-story without Basement	26	1st Floor		1	1
Total	383				

TABLE 3. EQUATIONS FOR PREDICTING ANNUAL MEAN CONCENTRATIONS FOR DIFFERENT TYPES OF HOUSES

Type of House	Prediction Equation	Correlation (X, Y)	$\hat{\sigma}$
One-story with Basement	$\hat{Y} = 0.40 + 0.60X$ (0.19)* (0.04)	0.82	0.53
Two-story with Basement	$\hat{Y} = 0.41 + 0.50X$ (0.16) (0.05)	0.79	0.31
One-story without Basement	$\hat{Y} = 0.59 + 0.48X$ (0.15) (0.05)	0.82	0.41
Two-story without Basement	$\hat{Y} = 0.42 + 0.45X$ (0.33) (0.09)	0.79	0.54
All Houses	$\hat{Y} = 0.42 + 0.56X$ (0.10) (0.02)	0.82	0.47

* (Standard error of parametric estimate.)

TABLE 4. PREDICTED LONG-TERM CONCENTRATION AND ASSOCIATED 95 PERCENT CONFIDENCE LIMITS FOR AN INDIVIDUAL HOUSE WITH A SPECIFIED SCREENING MEASUREMENT

Screening Measurement	Long-term Predicted Value	95 Percent Confidence Limits: Lower	Upper
1	1.0	0.0	3.7
2	1.5	0.1	4.7
3	2.1	0.3	5.6
4	2.6	0.5	6.5
5	3.2	0.7	7.4
6	3.8	1.0	8.2
7	4.3	1.3	9.0
8	4.9	1.6	9.8
9	5.4	2.0	10.6
10	6.0	2.3	11.4
12	7.1	3.0	12.9
14	8.2	3.7	14.4
16	9.3	4.5	15.8
18	10.4	5.3	17.3
20	11.5	6.1	18.7
25	14.3	8.1	22.2
30	17.1	10.2	25.7
35	19.9	12.4	29.1

Note: Predicted values are based on this limited data set and this specific type of statistical analysis. It is one of a series of studies being undertaken

TABLE 5. NUMBER OF FALSE POSITIVE AND FALSE NEGATIVE ERRORS IN THE STUDY SAMPLE USING 4 PCI/L AS THE CRITICAL VALUE

		Number of 2-day Measurements <4 pCi/L	Number of 2-day Measurements ≥4 pCi/L	Total
Number of 1-Year Measurements	<4 pCi/L	343	84 (false positive)	427
	≥4 pCi/L	11 (false negative)	90	101
Total		354	174	528*

* This includes those houses with screening measurements <1 pCi/L and ≥35 pCi/L which were excluded in the analysis of short-term versus long-term relationships.

CORRELATION BETWEEN SHORT-TERM AND LONG-TERM INDOOR RADON CONCENTRATIONS IN FLORIDA HOUSES

by: Ashley D. Williamson
Southern Research Institute
Birmingham, AL 35205

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Session B-III:

QA/QC of Measurement—POSTERS

AN E-RPISU (ELECTRET RADON PROGENY INTEGRATING SAMPLING UNIT) A NEW INSTRUMENT FOR MEASUREMENT OF RADON PROGENY CONCENTRATION IN AIR

by: P. Kotrappa, J. C. Dempsey, L. R. Stieff and R. W. Ramsey
Rad Elec Inc.
5330 J Spectrum Drive
Frederick, MD 21701 USA

ABSTRACT

A RPISU (radon progeny integrating sampling unit) is a generic name given to instruments which collect radon progeny on a filter paper and register the alpha radiation from the deposited progeny during the entire period of collection. An E-RPISU is a variation of this device with an electret located in an electret ion chamber serving as a sensor for registering the ions created by the alpha radiation. A conventional low-flow rate air sampling pump is used to collect the progeny on a one-inch diameter filter which is mounted on the side of the electret ion chamber (220 mL) such that the collected progeny ionizes the air inside the chamber. The negative ions are collected by a positively charged electret (0.127 mm thick) causing the electret voltage to drop. The electret voltage drop which occurs during the sampling period is proportional to the time integrated progeny concentration. The calibration factor for E-RPISU ranged from 1.2 to 1.6 V per mWL-day when sampled at 1 liter per minute. The calibration procedure and the performance of these devices are discussed. These devices have the sensitivity needed for indoor radon progeny measurements.

INTRODUCTION

It is known that the radiation dose to the lungs is almost entirely due to the inhalation of the particulate radon daughter products. These daughter products deposit in the airways of lungs and irradiate the basal cells of tracheobronchial and pulmonary epithelia. These cells receive dose not only from the deposited progeny but also from the alpha emitted by the daughter products formed after the deposition. This led to the concept of relating the inhalation hazard to the ultimate or the potential alpha energy concentration in working level units (WL). One WL is defined as the potential alpha energy concentration of the decay products of radon equivalent to 135,000 MeV per liter of air. Evans (1) has shown that one WL also corresponds to 16844 alpha from progeny contained in one liter of air.

The radon progeny integrating sampling unit (RPISU) is an instrument designed to measure the radon progeny concentration in WL units. These units collect the radon progeny on an air sampling filter paper. The alpha radiation is registered not only during the entire period of collection but also three hours after cessation of sampling. This data along with the volume of the sample collected can readily be converted into the progeny concentration in WL units. RPISU units have been reported which use thermoluminiscent dosimeters, alpha track detectors and solid state detectors. Their sensitivities are restricted by the fact that (a) the detectors are small and must be located close to the filter within the range of alpha radiation and (b) the area of filter paper is limited by the size of the detectors.

An electret is a piece of dielectric material carrying permanent electrical charge. Its effective surface charge decreases only by the collection of ions from the ambient air. An electret ion chamber is simply an electrically conducting plastic chamber containing a charged electret located at the bottom of the chamber. Electret ion chambers and their successful use as a passive environmental radon monitor has been described fully (2,3). An E-RPISU uses an electret ion chamber configuration to register the ions produced by the alpha radiation emitted by the progeny collected on the filter paper. The restrictions mentioned in the earlier paragraphs do not apply to these devices. The filter area could be large allowing a low cost pump to be used without worrying about the pressure drop. The detector (electret) need not be in the immediate vicinity of the filter. The chamber size can be relatively large.

The scientific basis of this type of RPISU was first established by Kotrappa(4). His goal was for determining the progeny concentrations in mine atmosphere during a short period of sampling. The key for the application of the devices for long duration integration required for indoor monitoring was the recent development of very stable electrets (2,3).

DESIGN FEATURES OF E-RPISU

Figure 1 shows the design features of an E-RPISU. A standard one-inch filter head is fitted to the side of a standard electret ion chamber (2,3). Small openings (2 mm diameter) are provided for air entry. A spring-loaded electret cover is used to cover the electret when the E-RPISU is either in storage or in transit. The left half of the figure shows the electret closed position and the right half shows the electret open position. The filter head is connected to a pump with a flowmeter. An alternate way is to use a pump with a calibrated critical orifice.

OPERATION OF E-RPISU

The initial surface voltage of the electret is measured using an electret surface potential voltmeter (2,3). It is fitted into the bottom of the chamber. The device is placed at a location where measurement has to be

done. Bring the E-RPISU chamber to the open position by unscrewing the top screw cap (Figure 1). Turn on the power to the air sampling pump. Note down the time and date of the start of the pump. Note down the air flow rate. After the desired period of sampling, bring the E-RPISU chamber to a closed position (Figure 1). Note down the final air flow rate, time and date. Stop the pump. The final surface voltage of the electret is measured. This can be done either at the place of measurement or after taking the device to the laboratory. The data to be noted are: (a) initial electret voltage, (b) final electret voltage, (c) air sampling duration, (d) the average flow rate.

E-RPISU™

Electret-Radon Progeny Integrating Sampling Unit (Schematic)

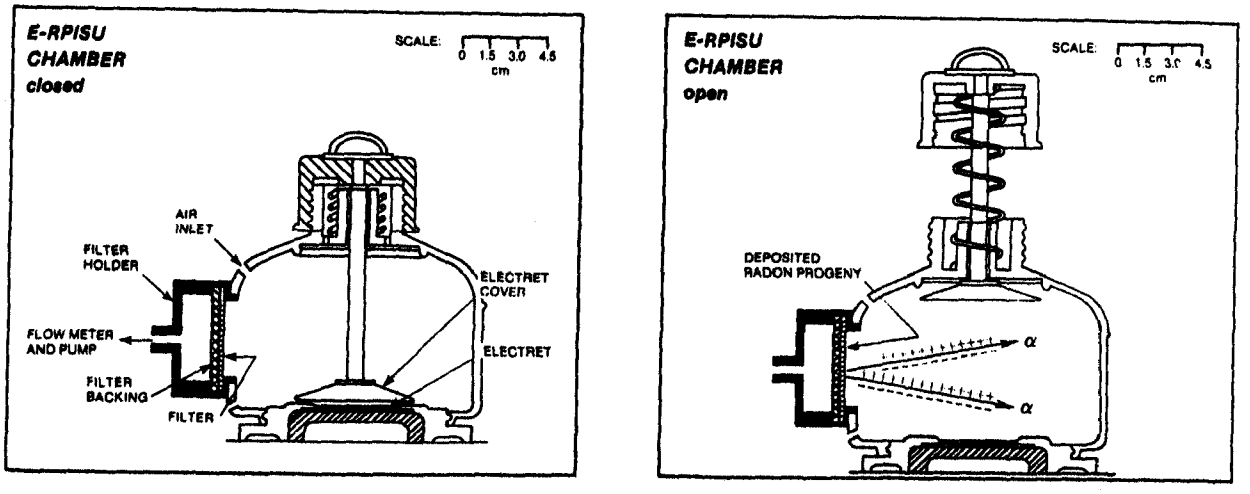

Figure 1. Schematic of the electret radon progeny integrating sampling unit.

CALIBRATION OF E-RPISU

The E-RPISU unit has to be calibrated in a standard chamber where the progeny concentration is precisely known. Most of the work was carried out in a well calibrated QC chamber (5). A set of three E-RPISU units were used, all of them with a starting initial electret voltage of about 750 V. These were run for a period of one day with a flow rate of 1 liter per minute at a progeny concentration of about 0.075 WL or 75 mWL. The final voltages of the electrets were recorded after allowing the registration of ions for an additional period of three hours after cessation of the sampling. Experiments were continued for several days until the final voltages of the electrets dropped to about 150 V. The calibration factors were determined using the following equation (1):

$$PC = (IV - FV) / (CF \times F \times D) \quad \text{------} \quad (1)$$

where PC is the progeny concentration in mWL
IV is the initial voltage of the electret
FV is the final voltage of the electret

F is the flow rate in liter per minute
D is the duration of sampling in units of days
CF is the calibration factor

Figure 2 shows such a calibration curve. The CF is on vertical axis and the average of the initial and final voltages of each measurement (also called the midpoint voltage or MPV) is on the horizontal axis. Equation (2) is a linear regression equation relating CF and MPV. The correlation coefficient is 0.89.

$$CF = 1.0744 + 0.000685 \times (MPV) \quad \text{---- (2)}$$

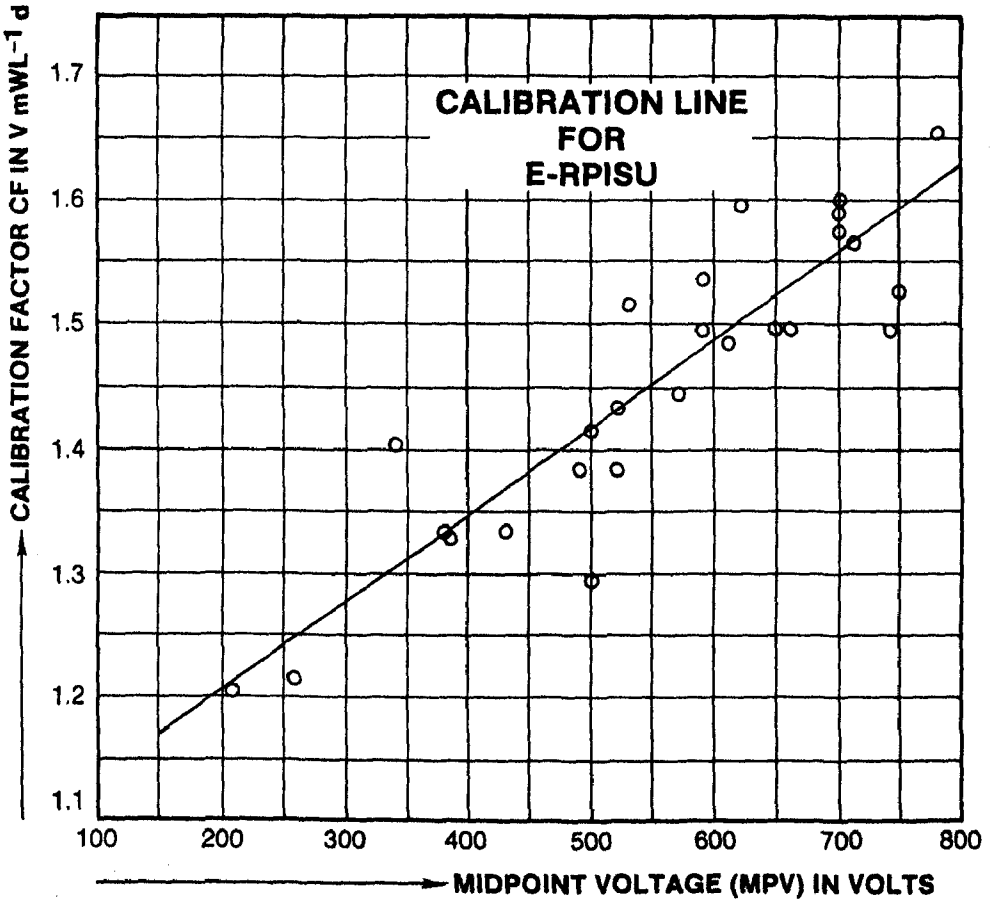

Figure 2. Calibration line for electret progeny integrating sampling unit. The air sampling rate is 1 liter per minute.

PROCEDURE FOR ACTUAL MEASUREMENT

In practice it is not always convenient or possible to wait an additional three hours after the cessation of the sampling for the registration of the alpha radiation of the tail end. It is possible to apply

a correction for this part. If DV is the voltage drop over a sampling period of DT days, then the average rate of change of voltage is DV/DT, which can be taken as constant and hold good at the end of sampling. Then the total expected tail-end voltage drop (TD) during the tail-end portion is given by equation (3).

$$TD = (DV)/(DT) \times \exp(- \quad \times t)\, dt$$

$$= (DV)/(DT) \times 0.03 \quad \text{-------} \quad (3)$$

where is the decay constant for the deposited radon progeny is about 33.3 per day. This is arrived at by assuming the half-life of the deposited progeny as about a half-hour or 0.0208 days.

Therefore, the total corrected voltage drop (CDV) is given by equation (4).

$$CDV = DV + (DV)/(DT) \times 0.03 \quad \text{----} \quad (4)$$

Let us assume the following values for a typical measurement:

IV=700 V; FV=650 V; DT=1.25 days; F=1.00 liter per minute.

Calculated parameters are:

MPV=675 V; CF=1.537; DV=50 V; CDV=51.2 V.

Result is:

$$PC = 51.2/(1.537 \times 1.25 \times 1.00) = 26.6 \text{ mWL}$$
$$= 0.0266 \text{ WL}$$

The average radon progeny concentration is 0.0266 WL.

Note that FV is the final voltage measured without waiting for the registration of tail-end ionization.

PERFORMANCE TEST

The performance of this device has been tested by the U.S. Environmental Protection Agency as part of the services provided by them for a new device. A unit was sent by mail to the EPA evaluation branch in Las Vegas and was evaluated at different progeny concentrations, at different humidities and at different condensation nuclei counts. The report concluded that the instrument gave a satisfactory performance and a measurement was never more than 12% off from the chamber value (6).

Two radon measuring companies entered the U.S. EPA (RMP6) program with these devices for single blind testing. The E-RPISU units were sent and received by mail. Measuring companies sent their results to the EPA where these results were compared to the chamber values. Table-1 gives these

results. The absolute value of relative errors were in the range of about 5%, much better than the EPA requirement of 25%.

As part of the international intercomparison study (blind test) conducted by the Department of Energy Environmental Measurements Laboratory (DOE-EML), four E-RPISU units were entered. Results are also shown in Table-1. As can be seen the absolute value of the relative errors were in the range of 2% to 3%.

TABLE-1. PERFORMANCE EVALUATION OF E-RPISU

Company Code	Detector Number	Measured Value (MI)	Target Value (TI)	Absolute Value of Relative Error [(MI-TI)/TI]
E-RPISU RESULTS OF SINGLE BLIND TEST CONDUCTED BY EPA-RMP-ROUND 6 (1989)*				
HJTLJ RP	793975	0.029	0.031	0.065
	793976	0.031	0.031	0.000
	793977	0.000	0.000	Blank
	793978	0.033	0.034	0.029
	793979	0.031	0.034	0.088
			Mean	0.0455
HFMEH RP	883570	0.030	0.031	0.032
	883571	0.045	0.047	0.043
	883572	0.049	0.044	0.114
	883573	0.000	0.000	Blank
			Mean	0.063
SINGLE BLIND TEST CONDUCTED BY DOE-EML (NEW YORK) AS PART OF INTERNATIONAL INTERCOMPARISON (1989)*				
RTI	571	0.136	0.137	0.007
	610	0.144	0.137	0.051
	547	0.135	0.137	0.015
	603	0.134	0.137	0.021
			Mean	0.024

* NOTE: These results are in the process of publication by the U.S. EPA and DOE-EML, respectively.

ERROR ANALYSIS

There are two sources of possible error. One is the random error associated with the dimensional variations, flow rate variations and other system errors. As seen from the performance (Table-1), this appears to be small and can be taken as about 5%. This was confirmed by running a large number of units in the same location. Another error is in the measurement of surface voltages. The measurement can be done only to within an accuracy of 1 V. When a difference between the two measurements is taken, as is always the case, the expected error is 1.42 V (square root of 2). This error can be substantial if the total measured voltage difference is small. Let us sample for one day at a concentration of 5 mWL at a flow rate of 1 liter per minute. The expected voltage drop is about 7.5 V, and the error in voltage measurement itself is 19%. This combined with the system error using the principle of quadrature comes out to be about 20%. This error can be reduced by sampling for a longer period or by sampling at a higher flow rate. Similarly at a concentration of 20 mWL, the error is about 7%.

SHORT DURATION MEASUREMENT

If one wishes to make a short-duration measurement -- less than one day-- then it is necessary to allow the tail-end portion of the ionization to be registered by allowing a delay of at least three hours. The correction equation (4) becomes less accurate.

It is possible to increase the sensitivity of the device by using a different electret of larger thickness (2,3). If another available 1.524 mm thick electret is used, the sensitivity is increased by a factor of 11. If a short duration sampling is desired at a low concentration, then this electret can be used in the same way. The CF will be 11 times the CF computed by using the equation (2).

DYNAMIC RANGE AND ECONOMICS

The electret can be used down to a surface voltage of about 150 V. The calibration factor does not hold good thereafter. The upper limit of the surface voltage is about 750 V (2,3) in the chamber used in this work. This leaves a headroom of 600 V. Therefore, an E-RPISU goes out of range when the total sample exceeds about 400 mWL-day. This corresponds to a sampling time of 20 days at a concentration of 20 mWL. In other words, one can use one electret for about 20 measurements in the case considered.

DISCUSSION

From the results of blind tests, it can be seen that the E-RPISU can be used to make an accurate measurement of radon progeny concentration. The units can be used as a mailable unit similar to the other passive monitors such as charcoal detectors or E-PERMs. Electret ion chambers are known to give good performance even at extreme temperatures and humidities encountered

in the indoor environment.

The electrets and the measuring device used with standard E-PERM radon monitors (2,3) are usable with the E-RPISU and hence, at a modest additional cost, it can be a useful addition to the laboratories already using E-PERMs.

The electrets have to be handled with some care which is described in our earlier work (2,3). Proper air flow calibration has to ensured as is the case with other RPISU units.

Kotrappa et al (4) have described how these devices can be used to measure both radon and thoron progeny concentrations. Further, a method is also presented (4) to measure the effective radon progeny concentration taking into account the contribution from the thoron progeny. That methodology is equally applicable to the present instrument.

ACKNOWLEDGEMENTS

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A REVIEW OF THE DETECTION TECHNOLOGY IN THE AT EASE RADON MONITOR

by: William E. Simon, MS
Sun Nuclear Corporation
Melbourne, Florida

Michael B. Schell
Residential Division
Honeywell, Inc.
Golden Valley, Minnesota

ABSTRACT

The AT EASE radon monitor is a relatively low cost device which utilities patented technology developed by Sun Nuclear Corporation under an SBIR contract with the U. S. Environmental Protection Agency. The device has been manufactured and distributed since August 1987. This paper is a description of the design and detection technique of AT EASE. At the heart of the device is a measurement circuit and pulse processing software which records PO-218 and PO-214 alpha events on the diffused junction surface of an inexpensive photovoltaic detector. The device has a nominal background counting rate of 0.1 counts per hour. A description of the noise elimination circuit is provided along with traces of alpha pulse shapes and pulse height spectra. Also defined is the display parameters and calibration technique. Test results on linearity, constancy, statistics, and diffusion time are provided.

A COMPREHENSIVE RADON ASSAY SYSTEM USING CELLULOSE NITRATE FILMS

by: Anders Damkjær & John L. Jørgensen
Department of Electrophysics
The Technical University of Denmark
DK-2800 Lyngby, Denmark

Milton E. McLain & Michael J. Shymanski
Department of Nuclear Engineering
Texas A&M University
College Station, Texas 77843-3133

ABSTRACT

A comprehensive radon assay system based on track production in cellulose nitrate films for the measurements of radon concentrations in air is described in detail. Specifically addressed are: 1) optimization of detector size and packaging vs. performance and processing considerations; 2) details of the chemical etching process for track visualization in exposed films; 3) microcomputer-based image analysis as applied to track counting; 4) effects of etch process parameters on track size distribution; 5) method for correction of observed track density for non-standard etch conditions; 6) radon-222 calibration procedures; 7) precision vs. radon concentration and exposure time considerations; 8) recommended quality assurance program; and 9) overall assay time/cost factors.

I INTRODUCTION

The awareness towards the population exposure from the presence of ^{222}Rn progeny in indoor air has led to a demand for simple and reliable radon assay systems. Most national surveys have found a log-normal distribution of the radon concentrations in dwellings (1),(2). This indicates that a small but not negligible fraction of the dwellings have radon concentrations so high that remedial actions should be taken. A large number of dwellings must have their radon concentration measured before the high level cases are found. Hence, the radon assay system must be a low cost system and yet it must yield a reliable estimate of the annual average radon concentration in order to avoid both false positives and false negatives (3).

Radon assay systems based on Solid State Nuclear Track Detectors are well suited for that purpose. A two month exposure time yields a sufficient number of alpha particle tracks for the radon concentration measurement. This exposure time is also necessary to average out the effect of changing meteorological conditions and daily variations in the radon concentration.

The radon assay system described in this work is based on cellulose nitrate film as the alpha particle track detector. The radon assay system has been used since 1984 for rutine measurements of radon in dwellings. A large number of measurements have been carried out on requests from private home owners and institutions and a survey, involving 200 radon dosimeters, has been made in a local area in Denmark (4).

II DETECTOR SYSTEM

DESIGN

The radon assay system is based on the closed dosimeter (detector) concept employing cellulose nitrate film,(Kodak LR 115 II), as the alpha track detector. Closed dosimeters have been proven superior to open dosimeters due to better protection of the alpha track detector during extended exposure periods as well as to a more well defined air quality inside the dosimeter (5),(6). The present assay system utilizes a cylindrical metallic container (figure 1), which provides adequate mechanical strength and shields the alpha track detector against infrared radiation. In addition, the metallic can ensures that electric fields due to static charges will not be present inside the dosimeter. Random static fields could affect the distribution of the radon progeny inside the dosimeter, and thus contribute to the observed data scatter.

A rectangular piece of the cellulose nitrate film, 1 cm x 2 cm, is fixed to the underside of the dosimeter's circular lid. Radon gas enters the dosimeter by passive diffusion via a 5 mm diameter hole in the lid, just beside the alpha track detector.

A filter paper membrane covers the diffusion hole. This ensures that radon daughters and pollutants in the form of aerosols will not enter the dosimeter and subsequently produce unwanted tracks or undesirable chemical reactions during the etch process.

Optimum dimensions for a cylindrical dosimeter depend on the efficiency function for the alpha track detector, i.e. the track forming efficiency as a function of alpha energy and angle of incidence (7). The dosimeter depicted in figure 1 yields a maximum efficiency for alpha particle detection from the decay of ^{222}Rn and ^{218}Po in the dosimeter volume, while still registering a large fraction of the alpha decays from plate out of ^{218}Po and ^{214}Po on the inside walls of the dosimeter.

The membrane covering the diffusion entrance hole of the dosimeter introduces a time constant for the establishment of radon concentration equilibrium between the interior of the dosimeter and the outside air. Standard laboratory filter paper has a diffusion constant for radon gas of the order of 0.002 cm^2/s. For the present dosimeter this corresponds to a time constant of 1.2 hours and an equilibrium radon concentration inside the dosimeter can which is 99 % of the concentration outside the dosimeter.

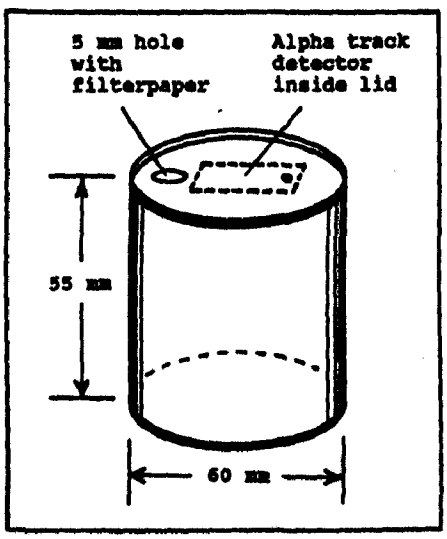

Figure 1. Radon dosimeter with Kodak LR 115 II as the alpha track detector.

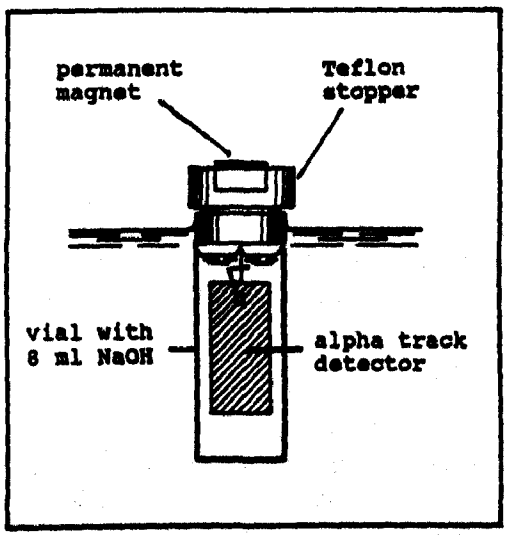

Figure 2. Etch cell with alpha track detector. The cell is submerged into a constant temperature water bath.

OPERATION

The detector material LR 115 II, is delivered from Kodak in boxes containing 25 sheets, each measuring 9 cm x 12 cm (Ref). Sheets of film are sandwiched between paper of low alpha activity in order to prevent damage to the sensitive layer and to maintain low background counts. For long time storage in the laboratory, (i.e. 1 - 2 years), the material is kept in its original box and stored in a refrigerator. Detector preparation starts with cutting the sheets into rectangles, i.e. 1 cm x 2 cm, without removing the protective paper. Again, this protects the sensitive surface against contamination which may affect the subsequent etch process. For mass production of similarly sized detectors, a machined punch die of the desired dimensions could be fabricated. However, for the laboratory-scale studies described here, a small cutting board designed for photografic paper is convenient. Next a 1.5 mm hole is punched in one end of the detector which will be used for holding the detector strip during the etch process. Then, the detector is mounted on the inside of the metal can lid with an asymmetric double-sided adhesive tape strip, which attachs firmly to the lid and less firmly to the detector. Finally, an identification number is scribed into the alpha track detector surface and the dosimeter can is closed.

If the dosimeter is to be exposed immediately after assembly, a can with a press-on friction-fit lid which is not completely radon-tight, is sufficient. However, if the dosimeter is to be stored for some period before use, a different type can is used which is sealed with a conventional can lid using a machine designed for that purpose. For these latter dosimeter cans, the membrane opening is covered with a radon-tight, aluminized Mylar pull-tape which is removed at the start of the exposure.

Public survey measurements, using the radon assay system described, can be carried out by sending the dosimeters through the postal system. A standard 20 cm long cardboard mailing tube will contain up to three dosimeters including instruction papers and return labels.

ETCHING

Following exposure and return to the laboratory, alpha track detectors are etched for 140 minutes in 2.5 N (10 %), analytical grade NaOH, at 60 degrees centigrade. This etch time is appropriate for LR 115 II which has been stored in the laboratory for more than half a year. However, it has been observed that freshly delivered detector material often requires up to 20 % longer etch time. A homogeneous etching is obtained when the etch bath is absolutely quiet i.e. without stirring and in absence of convection currents. In order to

achieve this condition, each detector is etched in an individual 10 ml glass vial submerged in a temperature controlled water bath. Figure 2 shows a vial with an alpha track detector in position for etching. The Teflon stopper fits loosely into the vial and the detector hangs freely on a stainless steel hook attached to the inside of the stopper. A small permanent magnet is fixed to the top of the stopper for easy handling of many detectors at one time, or if preferred, on an individual basis. After etching, the detctors are washed for 30 minutes in a stirred bath of demineralized water at 30 degrees centigrade, followed by a 2 minute, non-stirred bath of 50 % water/isopropanol. After a few minutes of drying in the air the detctors are ready for track counting.

TRACK COUNTING

The alpha track detector LR 115 II is well suited for automatic track counting with image analysis systems. The dark red dyed cellulose nitrate material comprises a 12 micrometer thick layer bound to a 100 micrometer thick, colorless transparent polyester film backing. This provides desirable mechanical strength for handling. After etching, a large fraction of the alpha tracks have penetrated completely the red dyed cellulose nitrate sensitive layer. When viewed under a transmission optical microscope, these tracks appear as bright spots on a red background with diameters up to 20 micrometers.

The track counting equipment consists of a microscope with a CCD video camera, a "framegrabber" board (video image digitizer) and a personal computer. The camera is a Philips type 56472 based on the CCD chip NXA 1011 which has 576 picture lines with 604 pixels each. Other CCD cameras with less resolution have also been employed. Several models of PC-based framegrabbers are now commercially available. The present radon assay system uses the "PC Vision - Plus" framegrabber from Imaging Technology, Inc. or a multipurpose framegrabber designed and built by one of our laboratories.

A low magnification microscope is used as the optical setup. The objective is a Will Wetzlar 4/0.10. No eyepiece is used and the effective magnification is adjusted by selecting the proper distance between the objective lens and the CCD chip in the camera. The alpha track detector is held flat in a brass frame placed in the focal plane of the microscope. An ordinary incandescent lamp is sufficient as the light source for the microscope. However, an infrared filter excluding wavelength above 735 nm must be used to avoid saturation of the CCD - camera which is rather sensitive in the infrared region of the spectrum. In addition, the optical system is fitted with a green filter with transmission in the range of 515 nm to 565 nm. This filter, combined with the red dyed alpha track detector, renders the background area essentially black, creating

extraordinarily good contrast for the tracks in the video image.

The combined effect of a homogeneously etched alpha track detector and a very high contrast between the tracks and the background allows for a simple and fast track counting algorithm. First the digitized image is binarized with a fixed threshold. This is followed by a segmentation procedure where the tracks are found and the track sizes determined. The algorithm, which is written in the C language, counts the tracks in the field of view within a few seconds.

ETCH VARIATION CORRECTION

In spite of a well controlled etch procedure some fluctuations in the bulk etch rate within a batch of detectors are regularly observed. Such inconsistent etching can adversely affect the results of the radon assay due to induction of "non-standard" track hole size distributions. The degree of etch can be quantified by measurements of the thickness of the remaining layer of cellulose nitrate on each detector after etching. The thickness is determined by the transmission of light of short wavelength through the red detector film. The present radon assay system uses a photometer consisting of a mercury vapor light (Philips TL 4W/33 Xm8) as the light source and a photo resistor (CdS, type T 906021) as the light detector. Also, taking into account the previously mentioned need for adjustment of etch time with increasing detector film storage time, it is clear that the etch time itself is not a reliable reference parameter. Hence, for the present radon assay system, a reference photometer reading is chosen in order to define the standard etching. The radon exposure calibration factor for the assay system is determined as a function of the degree of etch, (i.e. the photometer reading), and individual calibration factors are used for the alpha track detectors according to their photometer reading.

CALIBRATION

Calibration of the alpha track radon assay system is accomplished with reference to a set of calibrated scintillation cells (Lucas flasks). The calibration system consists of:

1. A radon reservoir made of a 200 liter closed steel drum containing 26 kg. of finely ground uranium ore (400 ppm). The equilibrium radon concentration in the drum is approximately 100 kBq/m^3.

2. An exposure chamber, 5.8 liter cylindrical steel vessel with a radon tight lid (O-ring sealing) and two Swagelok self-closing quick-disconnect fittings for attachment to the radon reservoir drum.

3. A set of calibrated scintillation cells.

A batch of up to 12 dosimeter cans is placed in the exposure chamber and the lid is tightly closed. The exposure chamber is connected to the radon reservoir and radon rich air is circulated for a few seconds using a small-capacity, diaphram-type air pump. The radon reservoir is then disconnected and a scintillation cell is connected to the exposure chamber. The radon-laden air is then circulated between the cell and the exposure chamber for about 3 minutes in order to homogenize the radon concentration. Finally, the exposure chamber is disconnected and the scintillation cell is counted to determine the initial radon concentration in the exposure chamber. During the next 4 hours a diffusion equilibrium is established in the exposure chamber between the radon concentration inside and outside the dosimeter cans. This makes the true start concentration for the exposure lower than that initially measured, and the appropriate correction for radon dilution by the can air volume must be calculated. After an exposure time of typically 3 days the vessel is opened and the dosimeters are again allowed to reach equilibrium before the alpha track detectors are removed for etching. The procedure described here typically leads to an exposure in the range of 500 kBq hours per cubic meter.

III RESULTS

Figure 3 shows the results of a calibration run involving 12 alpha track dosimeters. All dosimeters received a radon exposure of 1359 kBqhm^{-3}. The alpha track detectors were deliberately etched at progressively longer etch times ranging from 140 to 195 minutes in order to determine the relationship between the calibration factor K and the photometer reading (PMR).

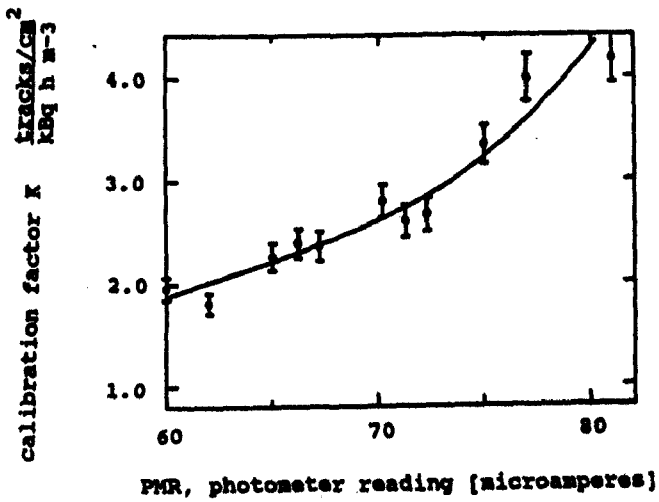

Figure 3. Calibration factor K versus etching (PMR)

Track counting was performed with a resampled picture field of 288 lines, each with 400 pixels. The area of the field viewed by the microscope was 0.0129 cm^2 and up to 10 fields were counted on each detector in order to obtain a minimum of 300 tracks observed per detector. The PMRs for the 11 detectors shown (one detector was over-etched) fall in the range from 60 to 81 microamperes. The measured calibration factor K as a function of PMR is approximately linear in the low etch region from 60 to 70 μA. In the high etch region from 70 to 80 μA, the slope increases, and above 80 μA the detectors begin to show signs of over-etching, i.e. the red dyed layer of cellulose nitrate may completely dissolve or separate from the colorless substrate film.

The reference etching for the alpha track detectors for the radon assay system is defined as the etching which gives a PMR of 64 μA. The current value for the calibration factor for the present setup is K = 2.16 [(tracks/cm^2)/(kBqhm^{-3})] and the sensitivty, (i.e. the slope of the calibration curve at PMR = 64 μA), is found to be S = 3.4 [%/μA]. As evident from figure 3 the calibration curve has no plateau on which the calibration factor is independent of the degree of etch. This is further illustrated in figure 4 which shows the track hole size distribution as measured by the present setup. The distribution for the group of 5 detectors in the low etch region (60 - 70 μA) has been compared with the distribution for the similar group in the high etch region (70 - 80 μA). Within each group the data have been averaged in order to obtain sufficient statistics.

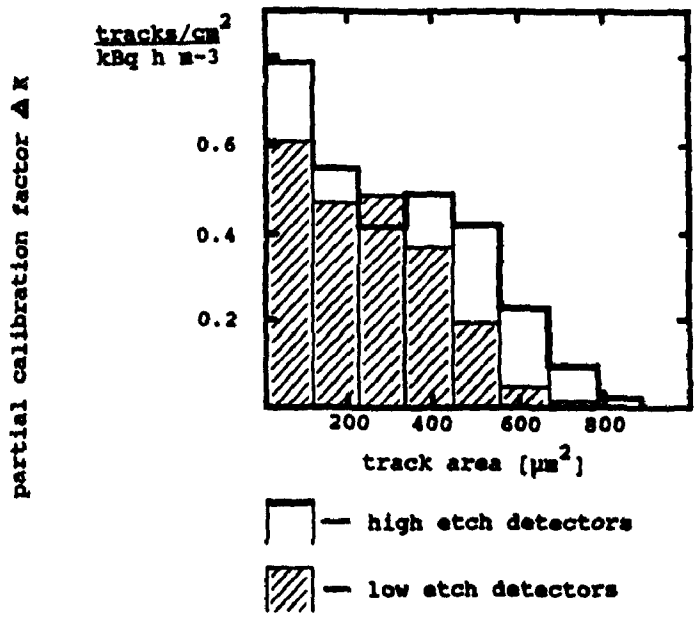

Figure 4. Track size distribution for low etch detectors compared with the distribution for high etch detectors.

The results depicted in figure 4 are expressed as the contribution to the calibration factor from tracks in the specified size region. The two distributions have similar shapes with indications of a local maximum around the average track size. The high etch distribution is shifted towards larger track sizes compared to the low etch distribution and in both cases the largest contribution comes from the smallest tracks. Hence, a prolonged etching not only increases the track sizes, it also continues to develop embryonic tracks into countable tracks.

FAST EXPOSURE FACILITY

The calibration procedure involving the exposure of dosimeter cans to radon gas in a closed vessel is a time consuming process. It is obviously more convenient to place the alpha track films in contact with an alpha active material in order to obtain an exposure of the track detectors. This exposure technique can not replace a radon calibration, however, it can serve as a fast and reliable check of the detector material and the etch process. In the present work we have used a piece of depleted metallic uranium for this purpose. The energies of the alpha particles emerging from metallic uranium are lower than those energies from radon and its progeny. This accounts for small discrepancies in the recorded track size distribution for the alpha track detector. Hence, the track forming efficiency as a function of the degree of etch must be separately established for the "uranium exposure" of the alpha track detectors. At the reference etching, (PMR = 64 μA), the track detector records alpha particles from metallic uranium at a rate similar to that of a track detector housed in a current dosimeter package placed in an ambient radon concentration of 54000 kBqm^{-3}.

INTERCOMPARISONS

The present radon assay system has participated in the CEC Radon Intercomparisons since 1984 (5),(6). This includes the Second (1984) and the Third (1987) intercomparison and "The First Small Scale" (1989) intercomparison. The radon assay system has also passed the US-EPA National Radon Measurement Proficiency Program, Test Round 6 (1989). The CEC Radon Intercomparisons have been carried out by The UK National Radiological Protection Board (NRPB) who has a 43 m^3 hermetically sealed environmental chamber at its disposal. In this chamber the ventilation rate, the humidity, the temperature and the aerosol composition of the air can be controlled. The ^{222}Rn concentration in the chamber can be maintained at constant levels ranging from 4 kBqm^{-3} to 15 kBqm^{-3}. In addition the NRPB has a "Fast Radon Exposure Device" (FRED) consisting of a 0.2 m^3 closed vessel where a radon concentration of 60 kBqm^{-3} can be maintained.

A total of 24 laboratories participated in the last two CEC Radon Intercomparisons of alpha track radon dosimeters. All participants mailed their dosimeters to NRPB for exposure and future return to their laboratories for evaluation of the NRPB exposure. Subsequently, the participating laboratories forwarded their results to NRPB who eventually revealed the radon exposures given to the dosimeters.

NRPB intercomparison results for the present radon assay system are presented in table 1. Several different exposure conditions were employed and equilibrium factors (F) from 0.24 to 0.88 were obtained. In addition to the test chamber exposures, two long term exposures in occupied dwellings were included in the intercomparison studies.

Evident from table 1, is the good agreement between exposures stated by the NRPB and those found by the radon assay system. Reproducibility of results by the current system was also found to be independent of the exposure conditions. This is further illustrated in figure 5 where the exposures as determined by the radon assay system are plotted versus the NRPB exposures. The deviations are of the order of 10 % with a maximum and minimum of 20 % and 3 %, respectively.

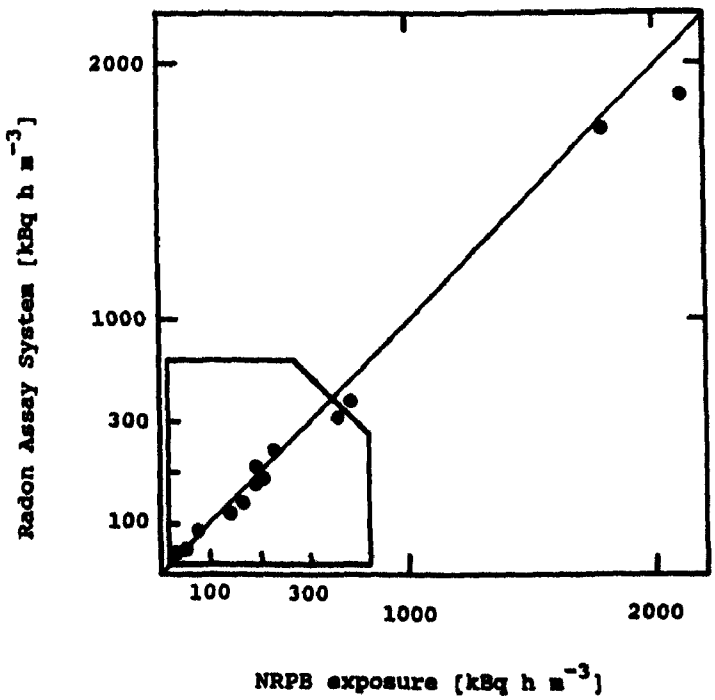

Figure 5. CEC Radon Intercomparisons. The results for the radon assay system is shown versus the NRPB exposures.

1984 NRPB radon exposure					Radon Assay System		
Run #	time [hours]	location	F	exposure [kBqhm^{-3}]	exposure [kBqhm^{-3}]	N	st.dev. [%]
1	1344	occp.house	0.41	147	118	10	11
2	41.25	env.chamb.	0.81	198	176	10	11
3	21.9	-/-		56	47	10	19
4a	2.0	FRED		82	84	1	
4b	3.75	-/-		171	140	1	
4c	7.75	-/-		360	303	1	
4d	16.33	-/-		771	674	1	
4e	31.75	-/-		1803	1742	1	

1987 NRPB radon exposure					Radon Assay System		
Run #	time [hours]	location	F	exposure [kBqhm^{-3}]	exposure [kBqhm^{-3}]	N	st.dev. [%]
1	1512	occp.house	0.44	197	210	10	13
2	3.42	env.chamb.	0.24	37	41	10	24
3	16.25	-/-	0.37	208	186	10	15
4	163	-/-	0.88	2120	1874	10	5

1989 NRPB radon exposure					Radon Assay System		
Run #	time [hours]	location	F	exposure [kBqhm^{-3}]	exposure [kBqhm^{-3}]	N	st.dev. [%]
1	96.8	env.chamb.	0.4	233	243	10	5

Table 1. The performance of the radon assay system in the CEC radon intercomparisons 1984 - 1989.

env.chamb. : NRPB's 43 m^3 radon environmental chamber.
FRED : NRPB's 0.2 m^3 Fast Radon Exposure Facility.
F : Equilibrium factor.
N : Number of dosimeters in the batch

PRECISION

The CEC Radon Intercomparisons include 8 cases where a batch of 10 dosimeters has received the same radon exposure. This allows for an experimental determination of the standard deviation for the radon assay system at the various levels of exposure. Final results are shown in the last column of table 1.

An analysis of the effect of the photometer correction has been carried out on the data from the 1987 and the 1989 CEC Intercalibrations. All 10 dosimeters in each of the 5 groups received the same exposure and have all been maintained under equivalent etch conditions.

The exposures for all alpha track detectors were re-calculated using the single value for the calibration factor valid for the reference degree of etch (PMR = 64 μA). The experimental standard deviation within each group was then calculated and compared to the standard deviation shown in table 1 where individual, (PMR - corrected), calibration factors were used. The results, shown in figure 6, demonstrate the effect of the photometer correction. All the standard deviations are reduced to sizes comparable to the standard deviations expected from the track counting statistics alone (dashed line).

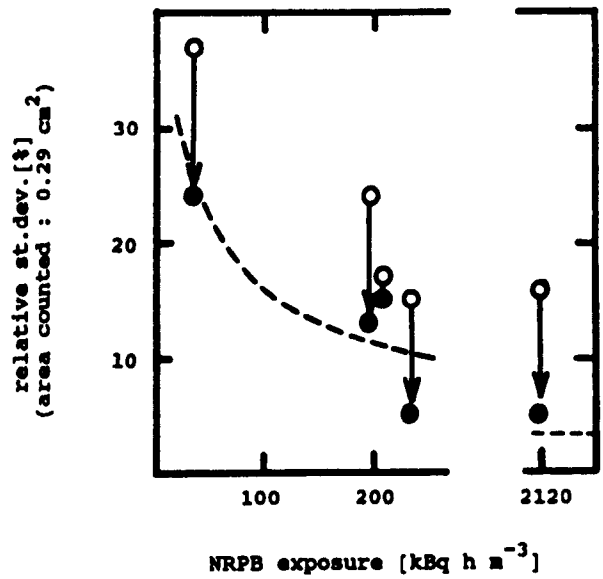

Figure 6. The effect of the photometer correction. The relative standard deviations for groups of 10 detectors are calculated with (●) and without (O) the photometer correction.

IV DISCUSSION AND CONCLUSIONS

The etch process is essential for the quality of the radon assay system. A homogeneous etching of the alpha track detector films is only obtained when the glass vials are kept thermally isolated so that convection currents in the etch bath are avoided. It is vital for the reproducibility of the etch process that analytical grade NaOH and demineralized water is used for the etch bath. All items in contact with the etch liquid must be kept clean. Also the track detector films must be kept free from fingerprints, dust or any unwanted chemicals. With these precautions, however, the radon assay system described in this work is simple to operate.

A standard radon measurement consists of a two months exposure. This is sufficient to average out the effect of meteorological changes as well as the daily variations in the radon concentration. A reliable estimate of the annual average of the radon concentration can then be made with a proper correction for the seasonal variation.

The precision (one standard deviation) obtained in a two months measurement is read from figure 6. At the level of 150 Bqm^{-3}, (i.e. 4 pC/l), the exposure will be 216 $kBqhm^{-3}$ and the radon assay system yields a precision better than 15 %. Similarly, a radon concentration level at 50 Bqm^{-3} can be measured with a precision better than 20 %.

The detection limit in a two months measurement is 10 Bqm^{-3}. This limit is determined as twice the standard deviation on the background measurements which include both the natural background of tracks on the alpha track detector and the radon exposure received during the transit through the postal system.

We estimate the cost in US$ for the etch equipment and the framegrabber to be $ 3500. The cost for each radon dosimeter, including materials and labor for both fabrication and processing, is estimated to be about $ 10.00. This is not based on a truly "mass production" scale of dosimeter processing and the individual cost could be reduced significantly when large groups of detectors are processed simultaneously.

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TWO-FILTER CONTINUOUS MONITOR FOR LOW LEVELS OF ^{220}RN AND ^{222}RN

by: D. Grumm and S. Schery
New Mexico Institute of Mining and Technology
Socorro, New Mexico

S. Whittlestone
Environmental Science Division
Australian Nuclear Science and Technology Organisation
Menai, New South Wales, Australia

ABSTRACT

A two-filter monitor system has been developed as a research instrument for measuring low-level atmospheric ^{220}Rn and ^{222}Rn. There is growing interest in thoron as a potential indoor air pollutant and measurement of both radon and thoron at low levels. The high sensitivities (typically 33.5 (α counts/hr)/(Bq/m^3) radon and 2.2 (α counts/hr)/(Bq/m^3) thoron for a two hour sample time) enable accurate time dependent measurements. We report results for collection and counting intervals of 2 hours and a sampling rate of 65 lpm into the 79 liter decay chamber. Continuous data taken 15 weeks at a site near Socorro, NM show the outdoor thoron concentration at 1m varying between 2 Bq/m^3 and 90 Bq/m^3 with the high typically in the early morning. Preliminary data suggest a significant correlation between outdoor thoron and radon, although there was measurable difference in response to temperature and rainfall.

INTRODUCTION

Measurement of low levels of radon and thoron is necessary for several reasons. It has been recommended that indoor levels of radon be reduced to its level outdoors, so concentrations of 1pCi/l (=37 Bq/m^3) and lower must be measurable. As concern over levels of indoor radon has increased, it is likely that thoron will also become more important as an indoor air pollutant. It has been estimated that the dose equivalent of thoron progeny is 20% that of radon progeny, and that there are 1,000 to 5,000 lung cancer deaths per year in the U.S. due to exposure to thoron progeny (1). Another reason is that the fundamental mechanisms responsible for thoron concentrations and their variations have not been investigated thoroughly. Outdoor levels are affected by meteorological and diurnal cycles (as are radon levels), but difficulty in measuring thoron due to its short half-life has hindered research in this area. The ability to measure thoron concentrations indoors will address the mitigation issue of whether soil or building material is the dominant source of thoron. Although thoron levels can be inferred from thoron daughter measurements, measuring thoron directly avoids the problem of uncertainties in the extent of equilibrium between thoron and its daughters.

The two-filter system described here is one of the few instruments in the world that can continuously monitor, unattended, low levels of radon and thoron simultaneously. A few other laboratories, such as the Elliot Lake Mining Laboratories and the Australian Radiation Lab have operational systems capable of measuring thoron gas, but designed for higher concentrations. In the research literature can be found descriptions of a few systems, some based on the ionization chamber principle, capable of low-level measurement but not operational today (see, for example, refs. (2), (3)). We report here results for collection and counting intervals of 2 hours, sampling outside air at a rate of 65 lpm.

THE TWO-FILTER METHOD

The system operates by the two-filter principle, which has been used to study airborne radioactivity for over twenty years. The basic theory is reviewed by Thomas and LeClare (4), so only a brief summary will be given here. The sampled air (which contains radon, thoron, and their progeny) enters the decay chamber through the entrance filter which removes the progeny. As the air travels through the chamber, the radon and thoron decay to their daughter products. The air that leaves the decay chamber passes through the exit filter, which removes those products that have been produced by decay of radon and thoron in the chamber. At the end of the sampling period, a detector counts the decays of these deposited progeny. Because this activity is only due to decays within the chamber, the radon and thoron concentrations in the entering air can be calculated from the decay equations. In fact, for a fixed sampling and counting protocol, the concentration of thoron and radon will be directly proportional to the decay counts from the respective daughters requiring only an experimental calibration.

EXPERIMENTAL DESCRIPTION

The air is supplied by a diaphragm pump (made by Gast) at a rate of about

65 lpm. The entrance filter, made by Gelman, is a type A/E glass filter. Just interior to the entrance filter are 10 closely-spaced layers of window screen, installed to reduce channeling and to produce a uniform flow down the length of the decay chamber. The pressure in the chamber is roughly 5 psi above atmospheric pressure, so any inflow due to leaks is minimized. The exit filter is a 5.7 cm wide roll of 1.2 μm acrylic copolymer Versapor filter made by Gelman.

The chamber is approximately 1 meter long, 35 cm in diameter, and has a volume of 79 liters. A Hewlett-Packard HP-71B microcomputer controls the entire two-filter system, including the operation of the pump, logging the detector output, and advancing the exit filter paper.

At the end of the sampling period, the area of the filter paper upon which the progeny are deposited is advanced to a silicon detector which counts the alpha decays of the progeny. An Amptek A-225 integrated circuit provides amplification of the signal. Simultaneously the next sample to be counted is collected. The microcomputer logs the detected α-decays in one of the following energy ranges: 4.7 to 6.4 MeV, 6.4 to 8.1 MeV, and 8.1 to 9.7 MeV from three electronic gates set on the output of the amplifier. The following progeny α-decays are recorded in these windows: ^{218}Po (6.0 MeV) and ^{212}Bi (6.05 and 6.09 MeV), ^{214}Po (7.7 MeV), and ^{212}Po (8.8 MeV). A multichannel analyzer (Canberra Series 10) was frequently used as a check on the microcomputer and to monitor gain drifts, which were typically 0.05% per hour. An alpha spectrum collected by the MCA for 2 hour sampling and 2 hour counting periods is shown in figure 1. From the number of events in the ^{214}Po peak, the radon concentration in the entering air is calculated to be 18 Bq/m^3 using the experimental calibration factor. Similarly the thoron concentration in the sampled air is 18 Bq/m^3 as calculated from the number of events in the ^{212}Po peak.

To calibrate the system for detecting radon daughter decays, air was first pumped through a column of uranium ore and sampled with a Lucas cell (with a calibration checked by the intercomparison program at the U.S. Environmental Measurement Laboratory) before entering the chamber. For a 2 hour sampling and 2 hour counting protocol, the sensitivity is 33.5 (counts/hr)/(Bq/m^3). For experimental calibration of thoron, a 66 Bq reference source of ^{228}Th was used which had been verified with a manual two-filter system that had been checked against equipment at Elliot Lake Mining Laboratories. The resulting sensitivity (for the same time intervals) is 2.2 (counts/hr)/(Bq/m^3). The background for both thoron and radon progeny is 0.5 (counts/hr).

RESULTS

To correlate outdoor radon and thoron levels with meteorological conditions, a microprocessor-controlled data acquisition system (Meteorite M800 by Weather Measure Corporation) continually logged barometric pressure, air temperature, wind speed, rainfall, and relative humidity at the sampling point. The air was sampled 1 meter above the ground and 5 meters outside the laboratory at the New Mexico Institute of Mining and Technology in Socorro, New Mexico. Several trends were apparent. Both the radon and thoron concentrations followed diurnal cycles; the averages for 70 days are shown in figures 2a and 2b. The radon levels follow an already well-established diurnal pattern (see, for example, ref. (3)), so we will focus on our thoron results. The typical uncertainty shown for the thoron levels are greater than that for radon due to

the decreased sensitivity. Because meteorological variables typically follow diurnal cycles, correlations with radon and thoron concentrations must be determined from data taken during intervals in which these variables do not follow their normal diurnal patterns. For the period 11-4-89 to 11-10-89, the pressure did not follow its usual cycle; the thoron concentrations for this period are shown as connected open circles in figure 3. These concentrations do not correlate with the pressure, but anticorrelate with the temperature. This can be explained by the same mechanism responsible for observed variations in radon concentration, which is already established (5) (6): warming of ground level air increases convection and mixing in the lower atmosphere, thereby reducing the concentration at 1 meter. This relationship can be visualized by modeling the thoron as a constant minus a term proportional to the temperature; this fit is shown as filled circles in figure 3. Increased wind also coincided with decreased concentrations, but in all cases the temperature was also high.

For both radon and thoron, rainfall coincided with decreased concentrations due to a decrease in flux. In figures 4a and 4b are shown the daily averages of the concentrations, with the time of rainfall denoted by vertical lines with lengths proportional to the amount of rainfall. On these days, the thoron level was 37% of that on dry days, and for radon, the level dropped to 75% of its dry-day average. This trend of a greater decrease in thoron level is expected for two reasons. First, a larger fraction of the soil volume emanating thoron will be blocked than the soil volume emanating radon due to thoron's shorter half-life. Second, because the detected thoron must have been emanated very close (meters, as opposed to kilometers for radon) to the sampling point, localized rainfall will have a greater effect on thoron concentrations. The correlation between thoron and radon levels can be seen from figure 5, in which all 2-hour levels are plotted; the linear correlation coefficient is 0.40.

The average concentrations for the data in figures 2 and 3 are for radon, 22.9 ± 2.2 Bq/m^3, and for thoron, 28.1 ± 3.7 Bq/m^3. Both values are representative of outdoor concentrations (refs. (5) (6)) although there is very little previous thoron data with which to compare. These data confirm an important point not commonly recognized: in terms of activity the concentration of thoron near ground levels outdoors is typically comparable with or higher than radon.

COMPARISON WITH THEORY

For a simple one-dimensional model of the atmosphere (5) with decay constant λ and constant eddy diffusion coefficient K, the concentration at a height Z above the earth can be written as:

$$C(Z) = \frac{J}{\sqrt{\lambda K}} e^{-Z\sqrt{\lambda/K}} \tag{1}$$

where J is the flux density. The steady-state flux density from homogeneous soil can be written in terms of soil parameters:

$$J = \gamma C(-\infty)\sqrt{\epsilon \lambda D} \tag{2}$$

where $C(-\infty)$ is the concentration at depth, ϵ is the soil porosity, D is the diffusion coefficient in the soil, and γ is a measure of the relative contributions of convective flow from a pressure gradient and diffusion. Specifically,

$$\gamma = \frac{V}{\sqrt{4\lambda\epsilon D}} + \left(\frac{V^2}{4\lambda\epsilon D} + 1\right)^{\frac{1}{2}} \tag{3}$$

where V is the Darcian flow velocity (assumed constant) through the soil. For typical flow velocities (of the order of 10^{-3} cm s^{-1}) at this site, $\gamma \leq 2$ for radon and γ is always very close to 1 for thoron. This explains the lack of correlations between thoron concentrations and pressure.

Considering the other terms in equation (1), the exponential varies only slightly with K for both radon and thoron. The eddy diffusion coefficient can vary between approximately 10 cm^2/s and 10^4 cm^2/s (2), depending on temperature-induced convection and wind speed. The dominant cause of variations in the concentration is K in the denominator. Using values of J, ϵ, D, and $C(-\infty)$ as measured by Schery et al (4), Table 1 shows the range of radon and thoron concentrations predicted by this range of K.

TABLE 1: COMPARISON OF PREDICTED AND OBSERVED CONCENTRATIONS

Gas	Predicted (Bq/m^3)	Observed (Bq/m^3)
Radon	11 → 74	3 → 37
Thoron	15 → 56	2 → 91

Considering the simplicity of the model and the uncertainty in k, the agreement is fairly good.

DISCUSSION AND CONCLUSION

1. We have successfully operated one of very few systems capable of low-level measurement of outdoor thoron.

2. Data at a Socorro, NM site show wide variation in outdoor thoron between 2 Bq/m^3 and 91 Bq/m^3, with a mean of about 28 Bq/m^3.

3. Thoron and radon are strongly correlated although the thoron concentration was more affected by rain and reached a maximum sooner, suggesting that thoron levels are more affected by local conditions.

4. The system should be useful for a number of research projects such as investigating the source of indoor thoron, determining typical

values and variation of outdoor and indoor thoron, studying factors affecting variation of thoron, and studying thoron in special locations (for example, maritime and highly urbanized areas).

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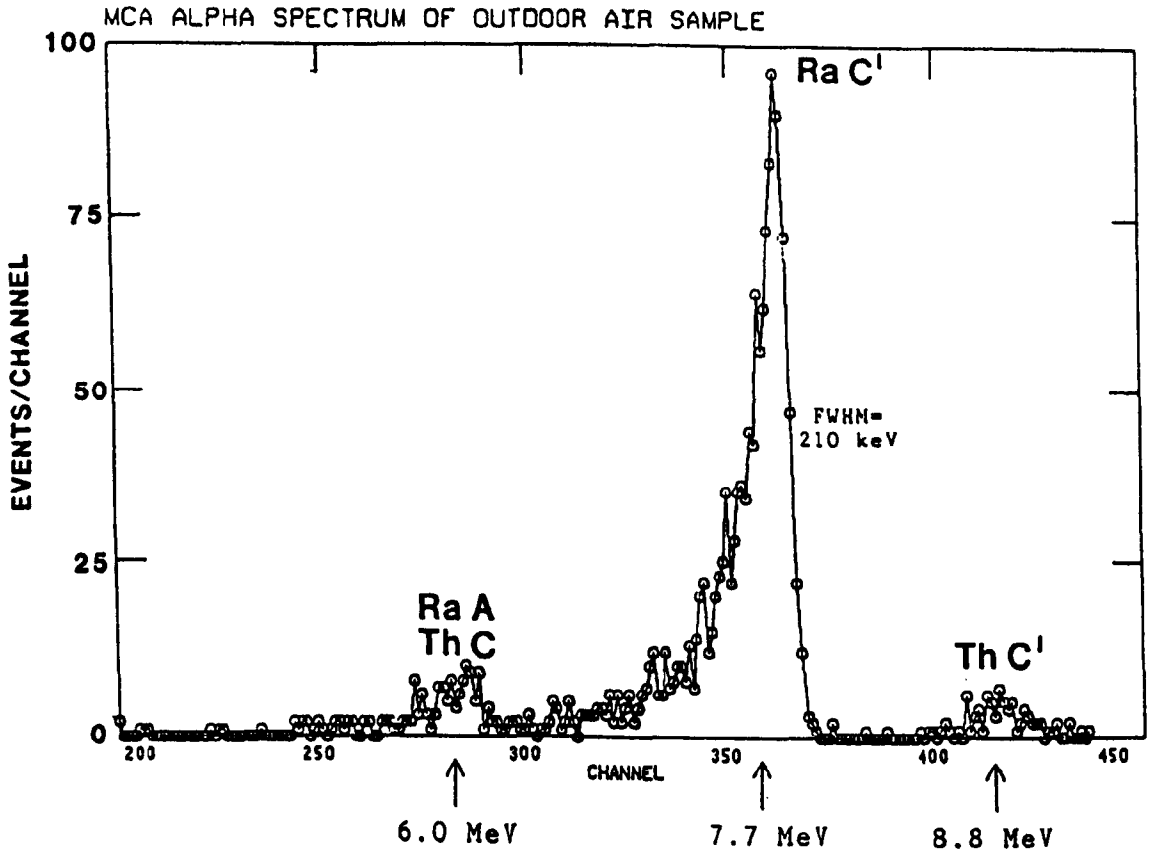

Figure 1. Typical alpha spectrum collected by multichannel analyser. Count rate in ThC' (^{212}Po) peak corresponds to 18 Bq/m^3 of thoron. Count rate in RaC' (^{214}Po) peak corresponds to 18 Bq/m^3 of radon.

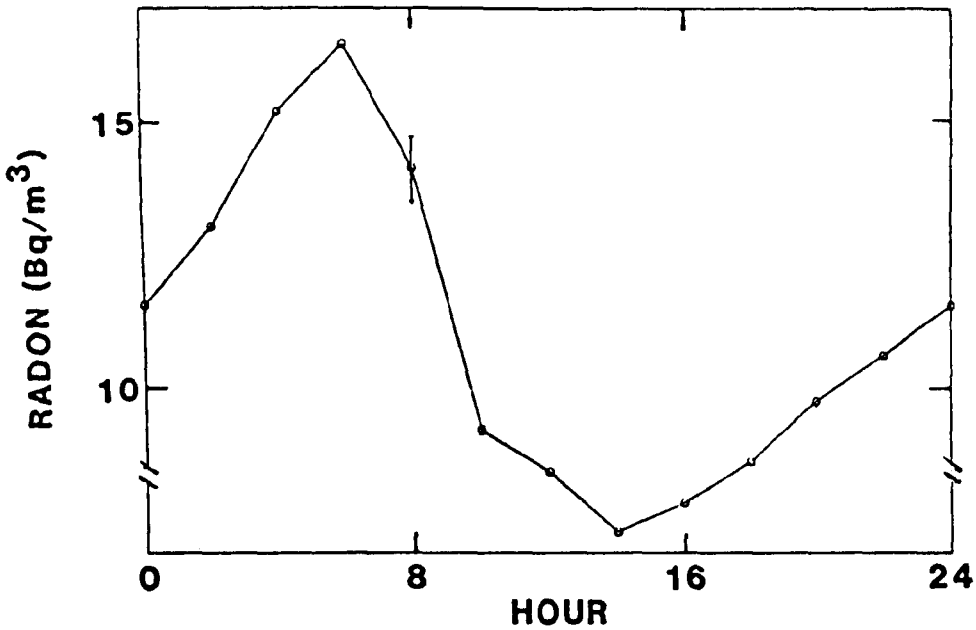

Figure 2a. Average diurnal cycle of radon concentrations based on data from 70 days, shown with a typical value of the uncertainty.

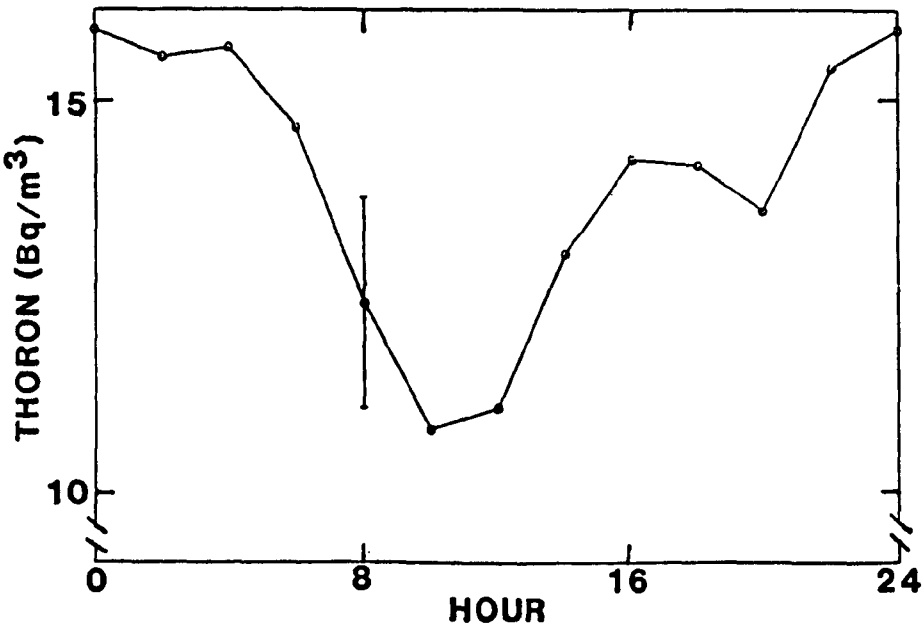

Figure 2b. Average diurnal cycle of thoron concentrations based on data from 70 days, shown with a typical value of the uncertainty.

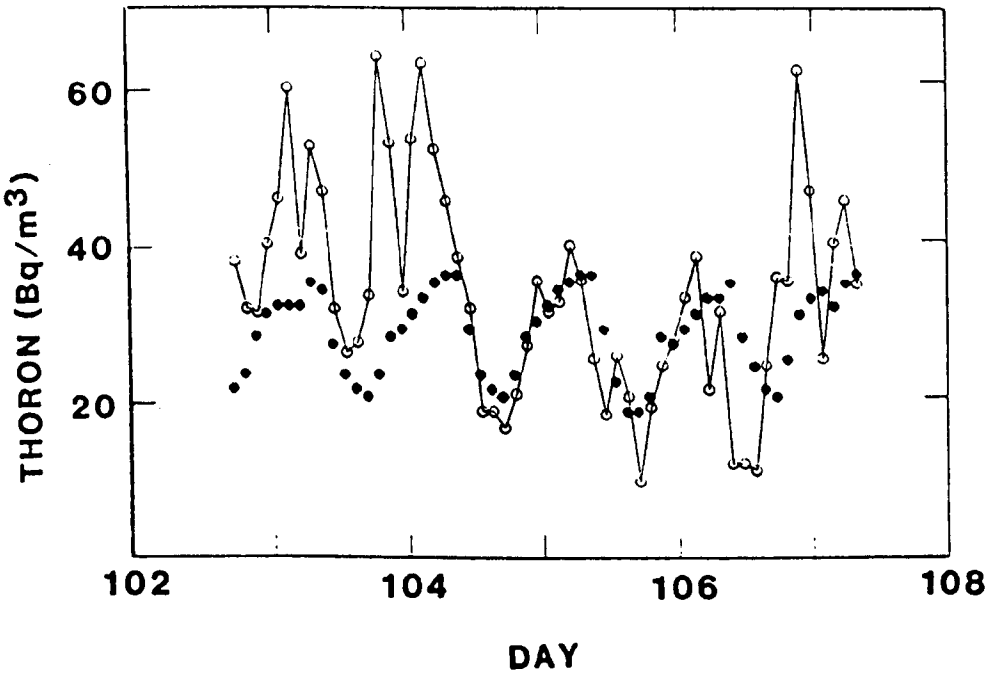

Figure 3. Measured thoron (connected open circles) for 11-4-89 to 11-10-89, and prediction (filled circles) based on phenomenological linear temperature dependence.

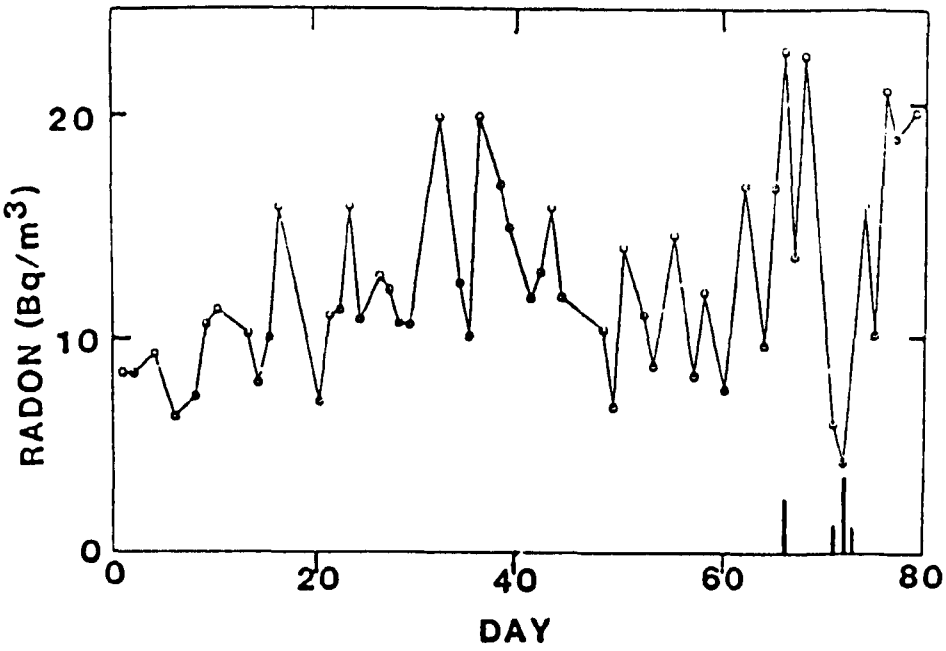

Figure 4a. Daily averages of radon concentrations 1 meter above Earth's surface from 7-23-89 to 10-13-89. Vertical lines denote rainfall, with the maximum equal to 0.9 cm.

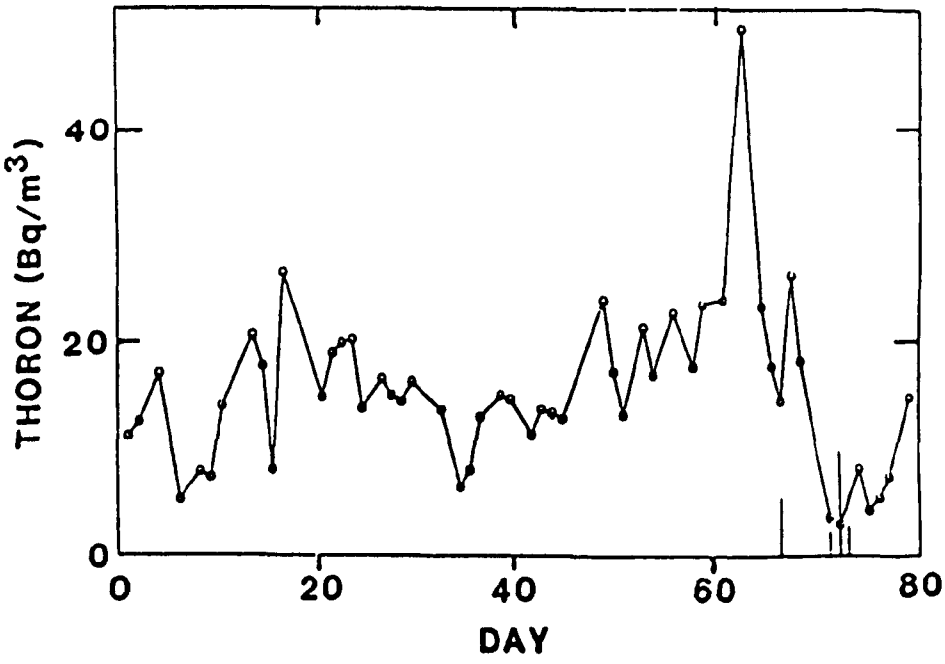

Figure 4b. Daily averages of thoron concentrations 1 meter above Earth's surface from 7-23-89 to 10-13-89. Vertical lines denote rainfall, with the maximum equal to 0.9 cm.

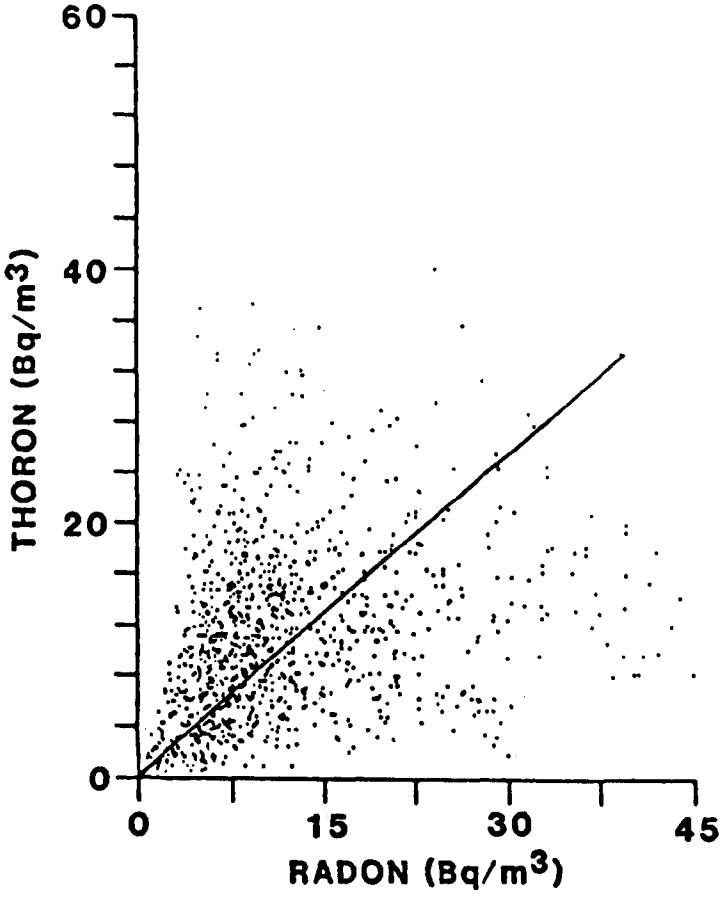

Figure 5. Plot of 2-hour thoron levels versus radon levels from 7-23-89 to 10-13-89

ACCURACY AND PRECISION OF PASSIVE LONG-TERM RADON DETECTORS AS A FUNCTION OF CONCENTRATION AND EXPOSURE TIME

Robert J. Lyon, Richard D. Hopper and Barry S. Parks,
U.S. Environmental Protection Agency, Office of Radiation Programs, Las Vegas, Nevada 89119

Mark Dickson and Michael Boyd, U.S. Environmental Protection Agency, Office of Radiation Programs, Washington, D.C. 20460

ABSTRACT

Four types of long-term passive radon detectors were exposed under three different exposure conditions to evaluate the effect of concentration and exposure time on accuracy and precision. The three sets of exposure conditions were designed to evaluate the effect of high concentrations (200 pCi/L) for short exposure periods (seven days) and low concentrations (0.4 to 60.0 pCi/L) for longer exposure periods (three and six months). The detectors were commercially available devices, with at least thirty of each type used for each set of exposure conditions.

Radon gas monitors (RGMs) were used as the standard measurement device for radon concentrations. At least five devices of each type were used for background monitoring. The detectors were sent to the manufacturers for processing in a routine manner.

The results of these exposures are presented with an analysis of precision and accuracy for the three sets of exposure conditions.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

Three types of alpha track detectors (ATDs) and one type of electret ion chamber (EIC) were exposed at selected concentrations and time periods to obtain estimates of precision and accuracy under different exposure conditions. Thirty to forty devices of each type were subjected to three different radon exposures.

Information regarding the accuracy and precision of long-term radon measurement devices is incomplete. This study will enable the U.S. Environmental Protection Agency (EPA) to better understand the benefits and limitations of these devices. This study may also indicate whether a brief, high radon concentration will produce the same results as an extended exposure to a low radon concentration.

MATERIALS AND METHODS

Two different types of radon detection methods were used in this study. The more common alpha-track method was used in three of the detector groups (designated as devices A, B, and C); while an electret was the detector media for Device D. Device A and C used CR-39 plastic, and device B used a cellulose nitrate film (LR-115).

The alpha-track detector (ATD) for measuring radon is based on alpha particles producing tracks on the detector media (Alter81, Lovett69). Alpha particles emitted by the radon decay products in the air impinge on the plastic or film, producing submicroscopic damage to the detector, commonly known as tracks. The detector media is placed in an etching bath of sodium hydroxide to enlarge the tracks. The tracks are accentuated when backlighted and viewed under a microscope or similar device. The number of tracks in a counted area are proportional to the concentration of radon to which the detectors have been exposed.

The electret ion chamber utilizes a charged detector (electret) to collect ions produced by radon and radon decay products (Kotrap88). The reduction in the charge of the electret over a known period of time is proportional to the integrated radon concentration. This voltage is measured on the electret before and after sampling with a portable surface potential voltmeter.

One group of detectors was exposed to 217 ± 7.2 pCi/L simultaneously for seven days for a total exposure of 1519 pCi/L-days. The radon chamber was controlled at 21°C and 50% humidity. Five to 10 detectors were unexposed to determine

background radon concentrations. Both the exposed and unexposed ATDs were sent to the appropriate manufacturers for normal processing. The EICs were analyzed by EPA staff at the Office of Radiation Programs (ORP) Las Vegas Facility using the manufacturer's methods and calibration factors to simulate routine analysis.

A second group of devices was exposed to naturally occurring radon concentrations for a three month period. The radon concentration (measured hourly by RGMs) varied from 0.4-60.0 pCi/L. The average concentration was 7.6 pCi/L for 98 days, for a total of 745 pCi/L-days. Both the exposed and unexposed ATDs were sent to the manufacturers as for the first group. The EIC results were obtained as described above. After the three month exposure, the same detectors were returned to the test facility since electret analysis is non-destructive.

The third group of detectors was exposed to naturally occuring radon concentrations for a 6-month period. The radon concentration averaged 6.2 pCi/L for 182 days for a total of 1130 pCi/L-days. Both exposed and unexposed ATDs were shipped to the manufacturers for processing. The EIC results were obtained after a 6-month exposure.

RESULTS AND DISCUSSION

Table 1 compares the known exposure levels and experimental standard error to the average measured values of each device.

TABLE 1. Known and Measured Radon Exposure (pCi/L-days)

Exposure Period (days)	Known (RGM) (X ± σ)	Measured Exposures (± 1σ) Device A (X ± σ)	Device B (X ± σ)	Device C (X ± σ)	Device D (X ± σ)
7	1520 ± 50	3040 ± 2900	1244 ± 369	1485 ± 87	1495 ± 96
98	746 ± 37	975 ± 347	600 ± 179	625 ± 36	763 ± 68
180	1135 ± 45	937 ± 258	1232 ± 244	909 ± 126	1100 ± 87

For convenience in determining accuracy and comparing different manufacturers at different exposure levels, the measured values were normalized by dividing by the known exposure in Table 2 and 3. Table 4 lists the accuracy and precision for devices A, B, C and D.

The results of the radon measurements are given in the appendix in Tables 5 through 8. The known values of the exposures were determined by calibrated radon gas monitors (RGMs).

TABLE 2. Normalized Radon Exposures of Devices A and B

Device A			Device B		
7 days	98 days	180 days	7 days	98 days	180 days
1.21	1.41	0.66	1.41	0.75	1.10
0.93	0.72	1.09	1.27	0.89	1.12
0.96	0.49	0.94	0.75	0.58	1.40
0.42	1.75	0.72	1.02	0.83	1.18
1.11	0.39	1.00	1.09	0.85	0.88
1.18	0.50	0.99	1.20	1.07	1.46
7.09	1.60	0.90	1.20	1.06	0.76
1.08	1.42	1.00	0.87	0.94	1.20
6.17	1.77	1.03	0.82	1.04	0.81
4.78	1.89	0.68	0.90	0.77	1.48
1.16	0.55	0.08	0.81	1.04	0.61
5.84	1.52	0.90	1.04	0.63	1.31
1.07	1.66	0.73	0.85	0.69	0.75
1.13	1.86	0.84	0.95	1.12	1.10
1.38	0.89	0.20	0.72	1.12	0.84
1.06	1.80	0.89	0.72	1.12	1.35
1.16	1.47	1.06	0.42	0.74	1.08
1.26	1.89	1.05	0.81	1.31	0.89
5.29	1.67	1.10	0.51	0.84	1.04
1.00	0.94	0.79	0.39	0.81	1.27
1.16	1.21	0.98	0.40	0.43	1.32
1.11	1.09	0.74	0.70	1.12	1.21
1.17	1.82	0.73	0.72	0.52	0.87
1.21	1.33	1.01	0.62	0.80	0.96
1.13	1.63	0.80	0.86	0.30	1.19
0.98	0.97	0.78	0.69	0.38	1.02
	1.44	0.82	0.71	0.71	1.05
	0.91	0.88	0.72	0.92	0.81
	1.42	0.72	0.74	0.95	0.92
		0.64	0.51	0.30	0.95
			0.93	0.96	1.11
			0.59	0.61	0.95
			0.44	0.56	1.14
			1.05	0.87	1.33
			0.93	0.54	0.98
			1.16	0.87	1.17
			0.77	1.03	1.40
			0.92	0.73	0.86
			0.73	0.61	1.36
				0.78	1.16

TABLE 3. Normalized Radon Exposures of Devices C and D

Device C			Device D		
7 days	98 days	180 days	7 days	98 days	180 days
1.00	0.83	0.78	0.95	1.04	0.95
0.96	0.80	0.88	1.08	1.03	1.00
0.98	0.89	0.90	0.98	1.01	0.96
1.00	0.83	0.78	0.95	1.04	0.95
0.96	0.80	0.88	1.08	1.03	1.00
0.98	0.89	0.90	0.98	1.01	0.96
0.96	0.89	0.79	0.99	1.04	1.00
0.97	0.88	0.72	0.95	1.02	0.96
1.03	0.83	0.67	0.97	1.02	0.98
1.01	0.80	0.83	0.99	1.01	0.98
1.05	0.81	0.89	0.94	1.01	0.96
1.03	0.94	0.84	0.99	0.93	0.87
1.00	0.83	0.81	1.00	0.98	0.94
1.05	0.80	0.61	0.95	1.00	0.95
0.97	0.82	0.60	0.93	1.04	0.99
0.88	0.92	0.89	0.95	0.95	0.90
0.96	0.83	0.88	0.97	1.02	0.97
1.01	0.84	0.61	0.94	1.09	1.03
1.03	0.86	0.80	0.96	1.11	1.01
0.94	0.77	0.89	1.03	0.98	0.92
0.98	0.84	0.79	1.00	1.00	0.96
1.00	0.90	0.87	0.95	0.94	0.90
1.01	0.88	0.85	0.99	0.99	0.97
0.93	0.87	0.86	0.98	0.97	0.93
1.05	0.89	0.93	0.94	1.01	0.97
1.04	0.82	0.89	1.15	1.46	1.33
0.89	0.85	0.55	1.22	0.93	0.87
0.97	0.90	0.66	0.94	0.97	0.92
0.91	0.84	0.84	0.94	1.02	0.97
0.89	0.87	0.89	0.94	1.04	0.95
0.98	0.89	0.84	0.94	1.05	1.01
0.88	0.90	0.66	0.98	1.03	0.98
1.03	0.78	0.85	1.00	1.02	0.98
0.85	0.88	0.86			
0.96	0.81	0.91			
0.87	0.85	0.74			
0.91	0.80	0.68			
1.01	0.77	0.80			
1.00	0.77	0.84			
1.09	0.81	0.89			
1.03	0.79	0.83			
1.02	0.75	0.86			
0.95	0.74	0.74			

TABLE 4. Accuracy and Precision for Devices A, B, C and D

Device/ Exposure Period (days)	Min.	Max.	Mean	Std. Dev. (σ)	% COV	% of Values Different from True Value by 25% or more
Device A						
7	.42	7.09	2.00	1.91	96	35
98	.39	1.89	1.31	.44	33	79
180	.08	1.10	.83	.23	27	33
Device B						
7	.39	1.41	.82	.24	30	65
98	.30	1.31	.80	.24	30	46
180	.61	1.48	1.09	.21	20	36
Device C						
7	.85	1.09	.98	.06	6	0
98	.74	.94	.84	.05	6	7
180	.48	.93	.79	.11	14	30
Device D						
7	.93	1.22	.98	.08	8	0
98	.93	1.46	1.02	.09	9	3
180	.87	1.33	.97	.08	8	3

CONCLUSIONS

Device A shows an improvement in precision as the exposure time increases from the 7-day to the 180-day exposure. The measured values of the 3-month exposure cannot be distinguished from the measured values of the 180-day exposure. Device A is biased high at low exposure times and biased low for longer exposures.

Device B shows no statistically significant change in precision as exposure time increases from 7 days to 3 months, but there is an improvement between 3 months and 6 months. The measured values of the 7-day exposure cannot be distinguished from the measured values of the 180-day exposure. Device B is biased low for short exposures and biased high for longer exposures.

Device C shows better precision than either A or B, but the precision decreases and the bias increases with increasing exposure time. Thirty percent of the measured values are more than 25% below the RGM values for 6-month exposures.

Device D, the EIC, showed superior precision and accuracy and retained the precision and accuracy as the exposure duration increased. In this study, device D was the most precise and accurate radon measuring device for long-term exposure conditions.

Research is continuing to determine the reason why, in these experiments, the ATDs manufactured with CR-39 appear to have an increasingly negative bias as the period of deployment increases. If there is some phenomenon that causes tracks to be lost over time, then it will be important to characterize this process and correct for it. The next phase of this project will cover a 12 month exposure period.

Appendix

TABLE 5. Measured Values of Radon by Device A

Radon (pCi/L d)			Background (pCi/L d)		
7 days	98 days	180 days	7 days	98 days	180 days
1838	1051	755	97	47	164
1410	540	1233	97	87	52
1456	363	1066	47	38	46
639	1305	815	161	25	39
1683	291	1140	77	30	15
1791	372	1123	89		10
10771	1192	1016			30
1637	1062	1136			36
9385	1321	1167			46
7255	1413	776			106
1761	408	90			
8875	1138	1026			
1622	1235	828			
1714	1386	956			
2100	666	230			
1606	1345	1009			
1761	1094	1201			
1915	1413	1188			
8042	1246	1251			
1525	624	900			
1761	900	1114			
1683	816	845			
1776	1359	833			
1838	990	1151			
1714	1213	909			
1490	726	885			
	1073	931			
	678	1000			
	1062	813			
		724			

TABLE 6. Measured Values of Radon by Device B

Radon (pCi/L d)			Background (pCi/L d)		
days	98 days	180 days	7 days	98 days	180 days
2146	557	1254			246
1931	662	1277			157
1140	429	1590			0
1543	616	1344	0.0 ± 65	0.0 ± 65	638
1663	637	997			0
1819	795	1658			29
1823	790	862			15
1322	700	1366			252
1247	778	918			378
1361	571	1680			216
1234	774	694			
1579	470	1490			
1285	515	851			
1445	835	1243			
1095	835	956			
1100	839	1529			
634	550	1220			
1225	980	1014			
770	627	1176			
591	604	1441			
613	319	1499			
1057	835	1382			
1092	385	985			
941	599	1088			
1312	223	1352			
1053	280	1161			
1090	531	1191			
1098	685	918			
1120	706	1044			
773	224	1080			
1419	720	1260			
899	454	1080			
664	420	1296			
1590	647	1512			
1413	403	1116			
1770	650	1332			
1165	770	1584			
1399	547	972			
1104	454	1548			
	582	1314			

TABLE 7. Measured Values of Radon by Device C

Radon (pCi/L d)			Background (pCi/L d)		
7 days	98 days	180 days	7 days	98 days	180 days
1521	622	884	18	18	44
1452	595	1004	18	12	30
1485	663	1018	18	15	33
1458	666	896	0	9	12
1479	658	816	15	6	38
1560	616	759	21	3	47
1533	597	941	18	6	27
1603	607	1011	9	9	53
1564	704	956	18	12	35
1521	617	919	27	6	53
1600	595	690			
1479	614	682			
1334	683	1006			
1458	619	1002			
1539	624	687			
1560	638	907			
1428	575	1014			
1497	624	891			
1524	668	991			
1539	656	966			
1419	648	979			
1594	665	1051			
1574	614	1009			
1360	631	622			
1468	670	749			
1381	629	954			
1357	651	1009			
1490	661	959			
1336	671	745			
1574	582	964			
1290	660	973			
1456	602	1029			
1321	631	838			
1387	599	770			
1532	572	941			
1517	576	542			
1659	606	1006			
1568	587	940			
1550	559	976			
1444	554	844			

TABLE 8. Measured Values of Radon by Device D

Radon (pCi/L d)			Background (pCi/L d)		
7 days	98 days	180 days	7 days	98 days	180 days
1448	774	1077	0 ± 6	0 ± 6	0 ± 6
1642	768	1136			
1485	751	1088			
1504	779	1134			
1440	760	1092			
1476	759	1107			
1506	750	1116			
1429	754	1090			
1503	692	986			
1520	734	1065			
1439	743	1074			
1415	775	1121			
1441	710	1021			
1469	761	1097			
1423	811	1165			
1453	825	1152			
1565	733	1039			
1516	743	1085			
1441	702	1023			
1504	740	1097			
1486	725	1057			
1434	754	1096			
1744	1092	1509			
1858	691	990			
1430	723	1048			
1427	761	1103			
1424	779	1083			
1427	782	1141			
1483	767	1108			
1515	760	1117			

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The EPA Diffusion Barrier Charcoal Adsorber for Radon Measurements in Indoor Air

David J. Gray and Sam T. Windham

Eastern Environmental Radiation Facility
U.S. Environmental Protection Agency
Montgomery, AL. 36109

ABSTRACT

An improved charcoal adsorber radon monitor capable of integrating radon concentrations from two through seven days has been developed by EPA. The EPA open-faced charcoal canister was modified by inserting a diffusion membrane on top of the activated carbon bed. This resulted in a decrease in the adsorption and desorption rates of the canister, as well as a decrease in moisture gain of the carbon. Excellent results have been obtained with this canister in environments where the radon concentration has varied by more than 10 to 1 and at humidity ranges from 20 to 85 percent. Radon measurements made with diffusion barrier canisters are less affected by temperature extremes and air movement than open-faced canisters.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer and administrative review policies and approved for presentation and publication.

INTRODUCTION

The open-faced passive charcoal adsorber used by the Environmental Protection Agency (EPA) and others in the radon industry responds quite well for a 2-day exposure period in most indoor situations, however, researchers have documented limitations of these devices under extreme conditions. Examples of these conditions include exposure in areas with relative humidity >60% (1), when radon concentrations vary by more than a factor of 10 (2), and measurements made with air movement in the vicinity of the adsorber (3) (4).

At the EPA's Eastern Environmental Radiation Facility (EERF) we have designed a diffusion barrier charcoal adsorber which responds more uniformly in extreme conditions and extends the integration measurement period. The EPA open-faced canister was modified by inserting between the retaining screen and the carbon bed a 1.25 mil polyethylene membrane containing twenty 0.091" diameter holes. This results in a decrease in the amount of water vapor and radon adsorbed by the carbon, a reduction in the rate of adsorption/desorption between the carbon and the environment, and improved integration capability. In this paper we will describe these improvements in radon measurement by the diffusion barrier adsorber.

RESULTS AND DISCUSSION

Radon Adsorption

Sets of 5 diffusion barrier canisters were exposed to the same radon concentration for periods of 1 to 10 days at low, medium, and high relative humidities in the radon chamber at the EERF. The chamber was operated in an "active" configuration thus presenting air flows of 10-30 LFM, not unlike conditions found in homes we

have measured. Three hours post exposure, the canisters were counted for 10 minutes using EERF SOP's (5). The collection efficiency of each canister, in terms of adsorbed radon on the carbon, was calculated from the following equation.

$$\text{Adsorbed radon (pCi/g per pCi/L)} = \frac{N}{E \times 70 \times Rn \times Df}$$

where:

N is the net count rate, (CPM),
E is the counting efficiency of the detector, (CPM pCi^{-1}),
Rn is the radon concentration during the exposure, ($pCiL^{-1}$),
Df is the decay factor for radon from the midpoint of exposure until the time of counting,
and
70 is the weight of carbon in the canister, (g).

The results for the diffusion barrier canister are shown in Figure 1. The collection efficiency has a positive slope for the first five days over the range of relative humidities and then begins to plateau, especially at high humidity for the remaining exposure period. This is in contrast to the collection efficiency of the open-faced canister at high relative humidities (>70%) (6) where the collection efficiency begins to decrease after an exposure period of 2 days. This difference is attributed to the characteristic of the diffusion barrier to reduce the adsorption of water vapor by the carbon. The break point of the carbon is the point where the amount of adsorbed water vapor is sufficient enough to prevent further adsorption and desorption of radon in the carbon bed (7). The diffusion barrier canister has not reached the break point of the carbon for up to 7 days at high relative humidities.

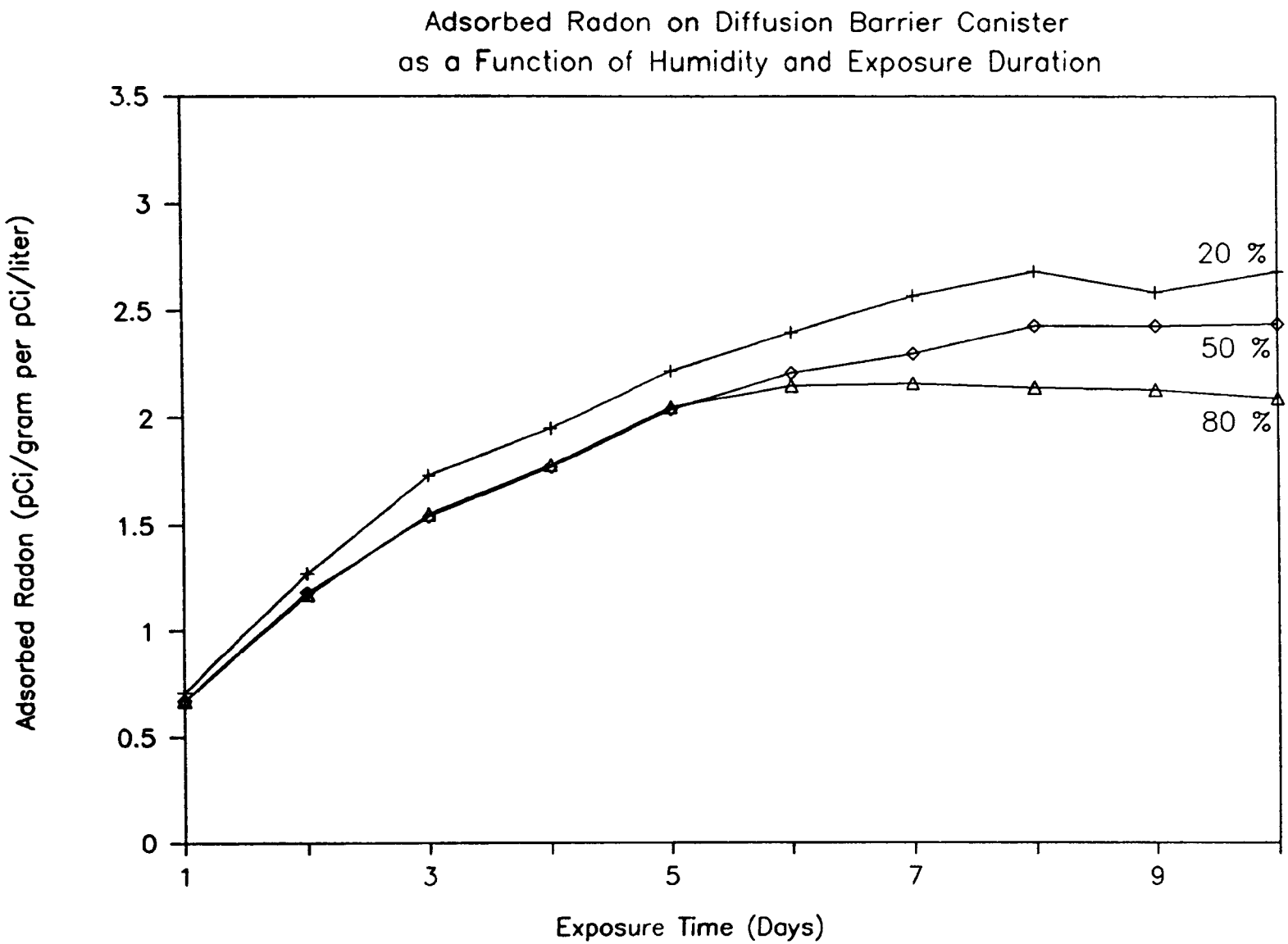

Figure 1. Diffusion Barrier Canister Radon Adsorption Curves

Radon Desorption

The effective half-life of radon in the charcoal canister is the combination of radioactive decay of the radon activity and desorption of radon from the carbon bed. To determine the effective half-life of radon in the diffusion barrier canister, a set of 5 canisters was exposed in the EERF chamber to 25 pCi/L for 7 days at a relative humidity of 50%. The canisters were sealed and counted to determine the radon activity (Q_0). They were then reopened and left to desorb in a low radon environment (<1 pCi/L) at a relative humidity of 40%. The canisters were resealed and counted periodically over 8 days post high radon exposure. The entire sequence was repeated at a relative humidity of 70%.

The data in Figure 2 shows that the effective half-life of radon in the diffusion barrier canister follows the equation $Q(t) = Q_0\ (t=0)\ e^{-\lambda t}$, where λ is the decay constant (hr-1) of the canister. The effective half-life of radon in the diffusion barrier canister is 61.5 hours (2.6 days) and 54.4 hours (2.3 days) at 40 and 70% relative humidities respectively. The small difference in desorption rate is attributed to the higher moisture content at 70% which accelerates the release of radon from the carbon bed.

The effective half-life of radon in the diffusion barrier canister is 3 times longer than that of the open-faced canister. In a desorption environment of 20 and 50% relative humidity the effective half-life of radon in the open-faced canister is 20.0 and 13.2 hours, respectively (6). The diffusion barrier greatly reduces the desorption rate of radon from the carbon, and in effect radioactive decay of radon becomes the predominant component of the effective half-life of radon in the diffusion barrier canister.

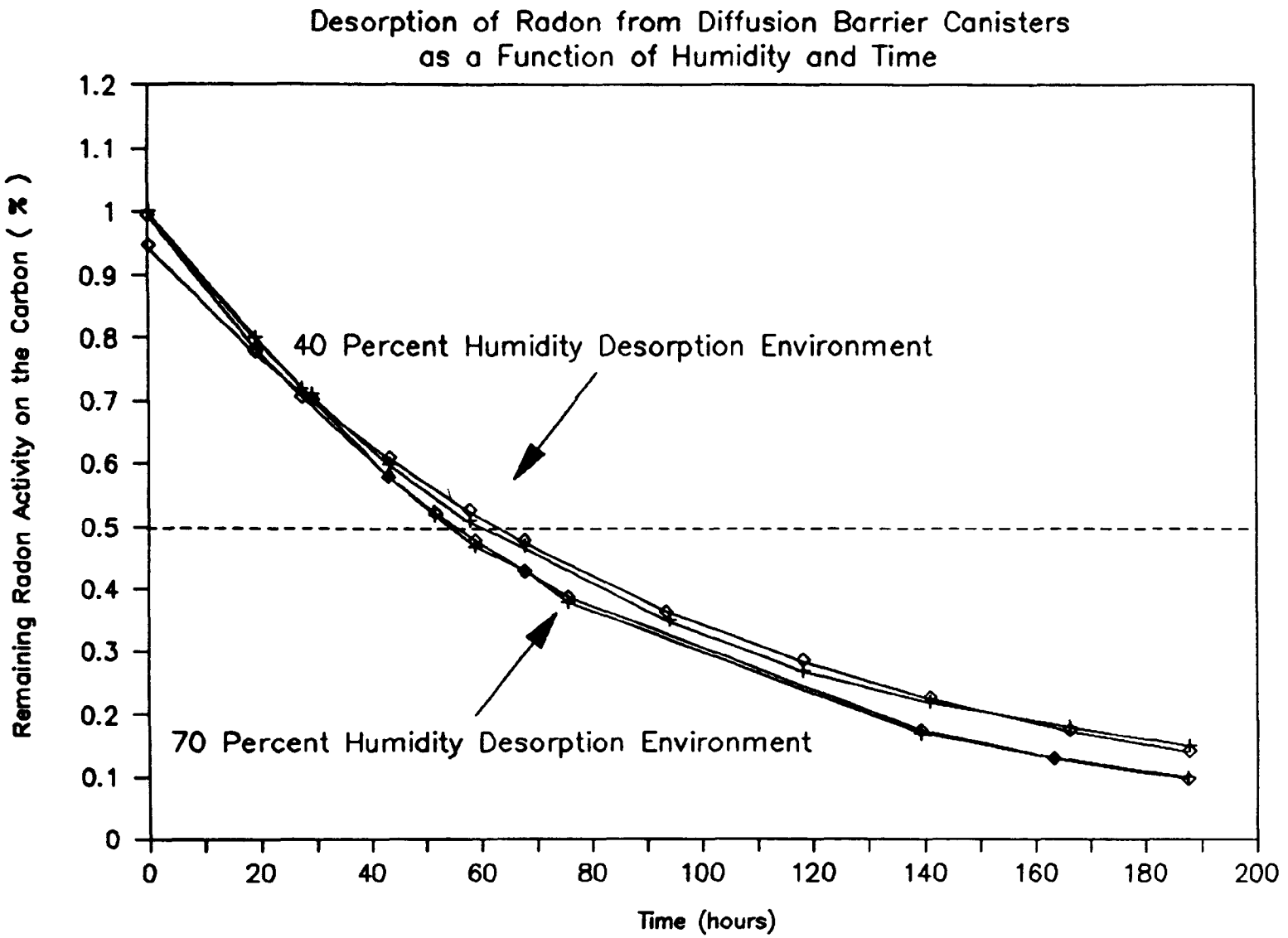

Figure 2. Diffusion Barrier Canister Desorption Curves

Response Of The Diffusion Barrier Canister During Variable Radon Concentration

The ability of diffusion barrier canisters to integrate time-weighted average radon concentrations was evaluated through a series of exposures. Sets of 5 canisters were exposed for periods of 2 and 7 days during which radon concentration varied from 6 to 1 to an extreme case of 100 to 1. The results of these exposures are shown in Figures 3 through 5.

In Figure 3, diffusion barrier and open-faced canisters were exposed at 70°F and 85% relative humidity for a period of 2 days. During the exposure period radon concentration varied from a low of 6 pCi/L to a high of 43 pCi/L. At the end of exposure the canisters were analyzed and the mean performance ratio (the mean of the reported radon concentration of the set of 5 canisters divided by the true average radon concentration of the continuous monitors) was calculated. The mean performance ratio for the open-faced and diffusion barrier canisters was 0.99 and 1.0 respectively, for the 2 day exposure period.

The result of diffusion barrier canisters exposed for 7 days at 70°F and 85% relative humidity under extreme radon variability is shown in Figure 4. The radon concentration varied from a low of 0.5 pCi/L to a high of 53 pCi/l during the period. The 5 canister mean performance ratio of 0.85 reflects the ability of the diffusion barrier canister to provide acceptable results under conditions of extreme radon variability.

The result of diffusion barrier canisters exposed for 7 days at 70°F and 50% relative humidity is shown in Figure 5. The variation in radon concentration was less extreme, varying from a low of 5 pCi/L to a high of 50 pCi/L. The mean performance ratio was 1.06 for the exposure period.

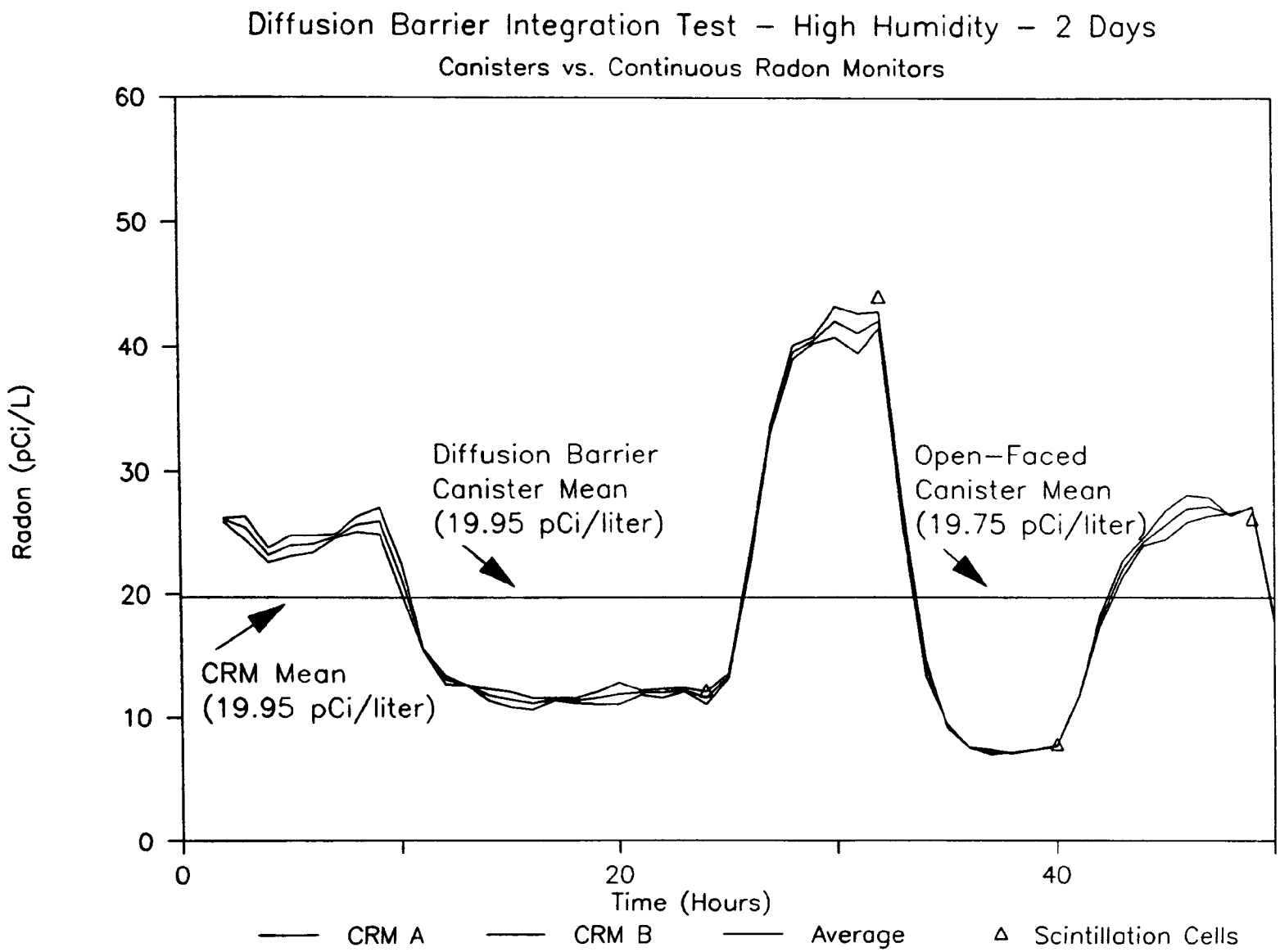

Figure 3. Diffusion Barrier and Open–Faced Canister Passive Chamber Test

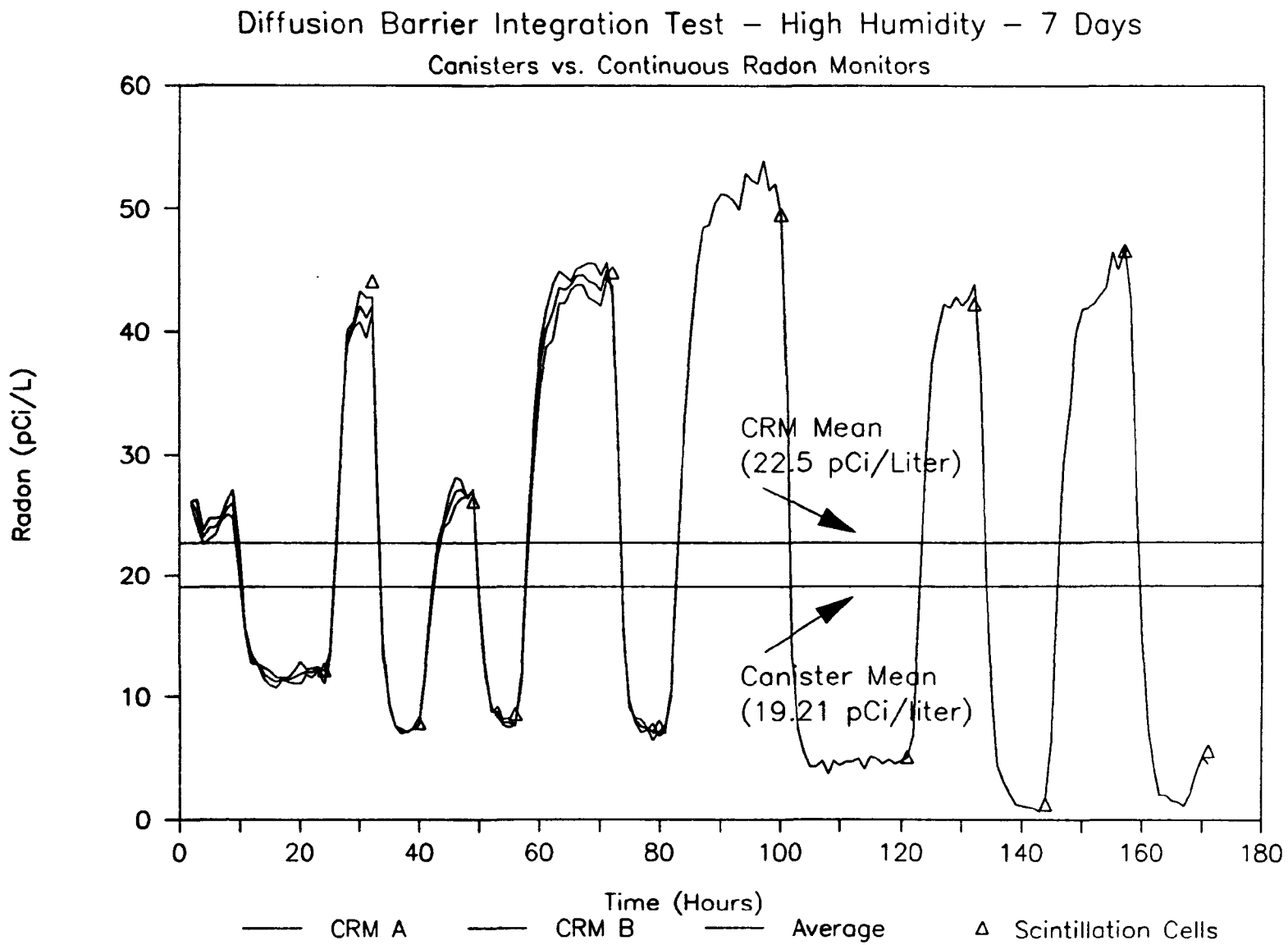

Figure 4. Diffusion Barrier Canister Passive Chamber Test

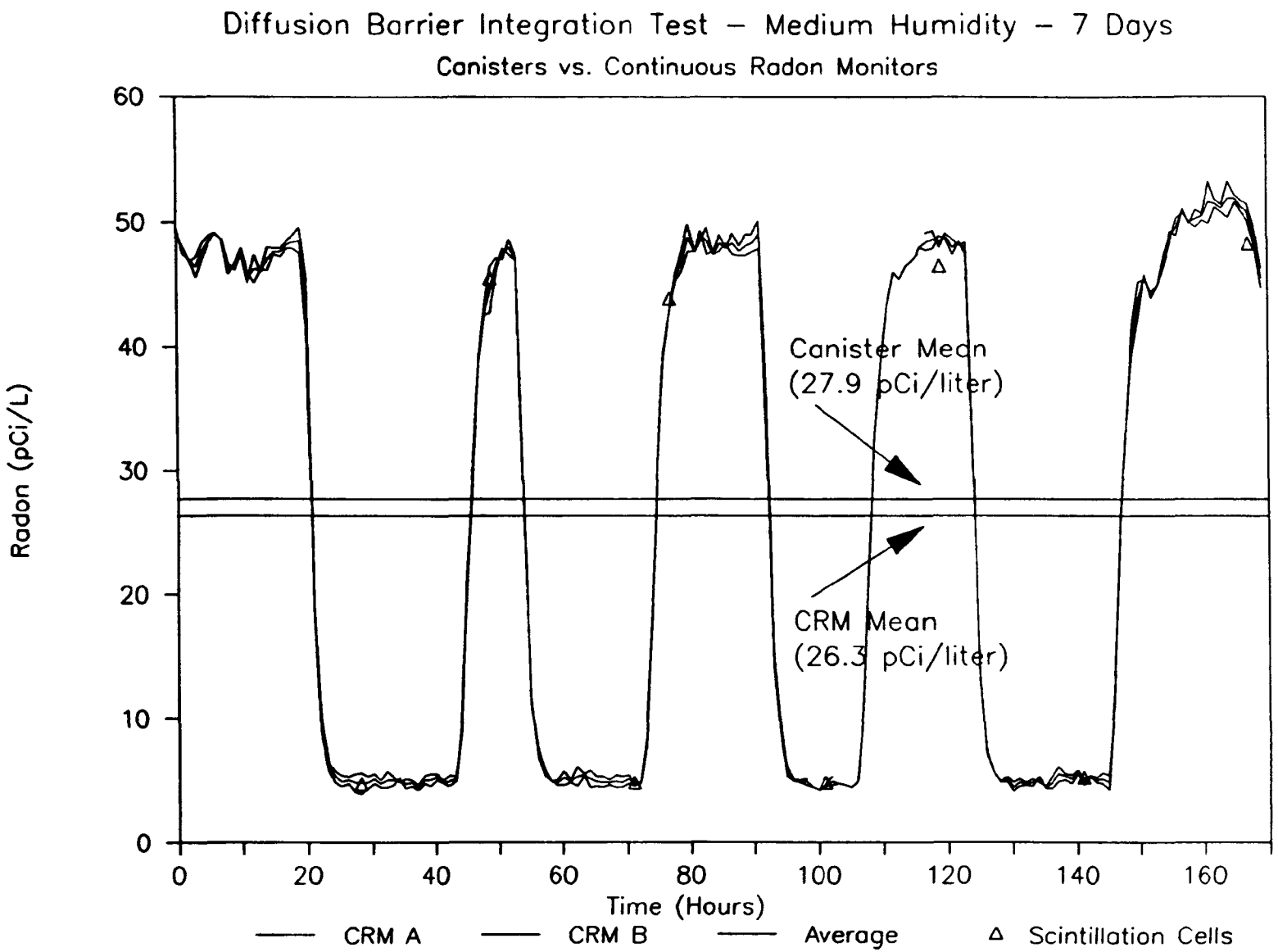

Figure 5. Diffusion Barrier Canister Passive Chamber Test

Field tests were also conducted in several residential structures throughout the country. Figure 6 shows the result of 5 diffusion barrier canisters exposed for 160 hours in the basement of a house in Michigan. The mean concentration of 7.3 pCi/L for the canisters compares quite well to the 7.2 pCi/L average measured by the continuous monitors.

CONCLUSIONS

Testing of the diffusion barrier canister at the EERF and in the field has shown that it can measure average radon concentration for an extended exposure period compared to the open-faced canister. The diffusion barrier decreases the amount of water vapor adsorbed by the carbon and reduces the adsorption/desorption rate of the carbon creating a more uniform sampling rate of the canister. This improves the retention and integration ability of the canister under extreme environmental conditions.

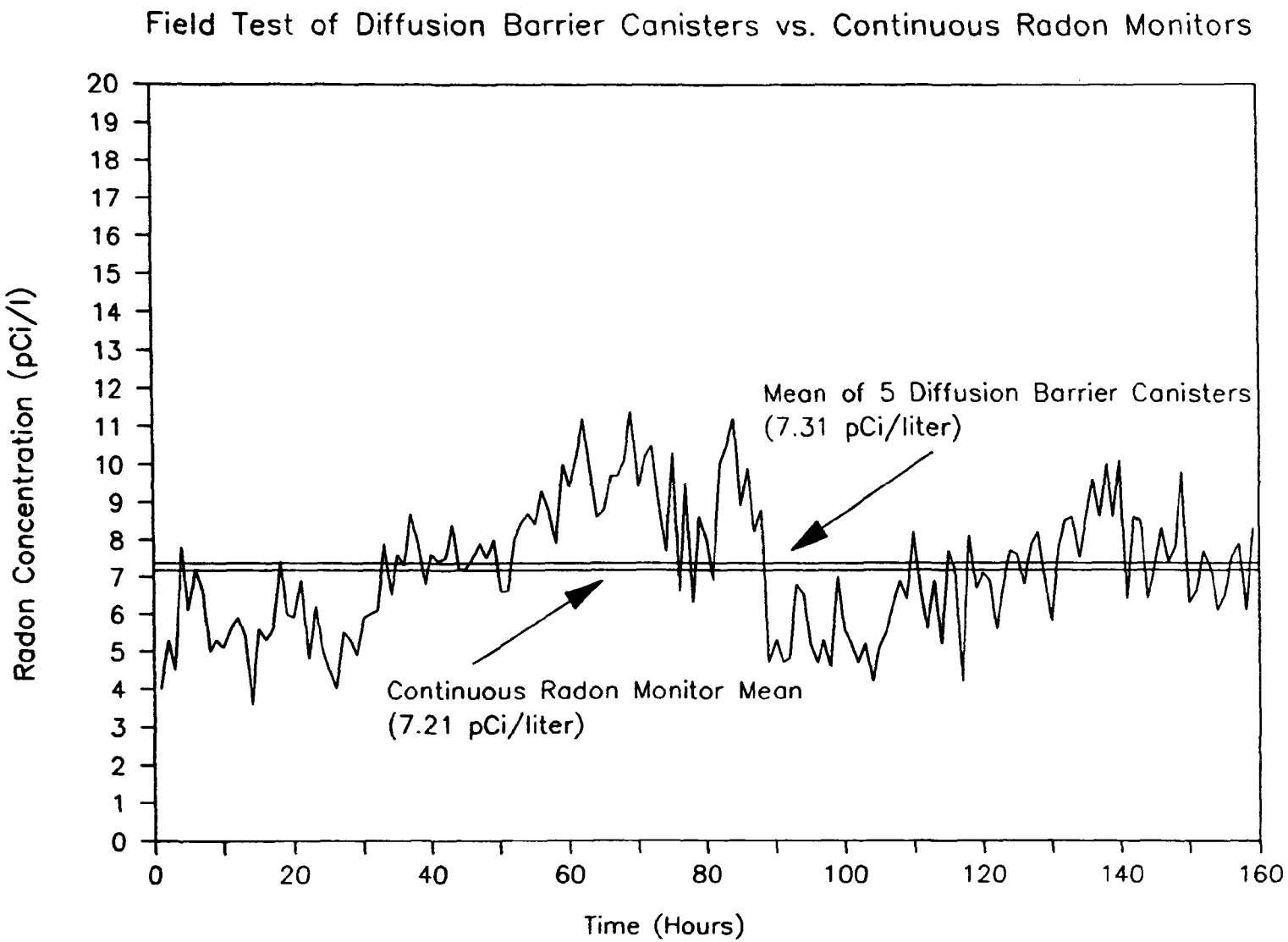

Figure 6. Diffusion Barrier Canister Field Test

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Session B-IV:

Short-/Long-term Radon Measurements—POSTERS

SURVEY OF RADON 222 IN MONROE COUNTY, PENNSYLVANIA

P. N. HOULE AND DOUGLAS N. BLICK
DEPARTMENT OF PHYSICS
EAST STROUDSBURG UNIVERSITY
EAST STROUDSBURG. PA

ABSTRACT

A high density of home radon concentration measurements over one Pennsylvania County was made. Agreement between these results and larger area but lower density measurements is found to be good. It is concluded that a measurement density of approximately .15 measurements per square mile will yield results in good agreement with a density of 1.39 measurements per square mile. It is also found that Pennsylvania, in part or taken as a whole, has three times as high a percentage of homes with radon concentrations greater than 4.0 Pci/L than present E. P. A. estimates indicate.

Survey of Radon 222 in Monroe County, Pennsylvania

P.N. Houle and Douglas N. Blick

Department of Physics, East Stroudsburg University, East Stroudsburg, Pennsylvania

Introduction

Numerous surveys of radon concentrations over relatively large geographic areas have been reported.[1,2,4] This report is of radon concentrations over a relatively small area. (Monroe County covers an area of approximately 601.4 square miles, and is in the northeast corner of Pennsylvania.) Eight hundred thirty-three (833) measurements were made yielding a measurement density of 1.39 measurements per square mile. The data reported herein covers all 23 Monroe County zip codes.

East Stroudsburg University has set up a radon testing program primarily for homes within its county--Monroe County--with the purpose of determining average radon concentrations. All persons interested in the program could write to the university to request test kits. These kits consisted of the now traditional charcoal canister with diffusion barrier, and were distributed for a price, at first, of $10.00, then one year later, $12.00.

The test results are represented by zip code area, as shown in Figures #1 and #2.

Materials and Methods

The E.S.U. Radon test kit consists of two metal canisters, each, 2 inches in diameter and 1.5 inches tall. Inside each canister are 15 grams of activated carbon-type PCB 12x30 made from coconut charcoal. The carbon is held in the canister by a metal screen, with mesh openings of 0.5mm. This metal screen, which is cut to fit inside the canister, is held securely against the charcoal by a bead of silicon caulk.

The lid of this canister has a 1/2-inch diameter hole through its center. A 1-inch square nylon mesh with openings of 30 microns and a thickness of 70 microns is glued to the bottom of the lid covering the hole. A 3-gram, drop-in style, sorb-it bag is taped to the bottom of the lid covering the nylon mesh and hole.

To make the canister lid secure and air tight, black electrical tape is wrapped around the perimeter of the canister cover along its seam. Duct tape is then placed over the hole on the lid, thus sealing the canister.

Both containers are weighed together to the nearest tenth of a gram before and after exposure to determine the moisture captured within the container. Instructions sent with the kits indicate the kits should be placed in the home

side by side. After exposure for seven days, the detectors are analyzed with a two-inch diameter sodium iodide crystal and associated electronics, including a multi-channel analyzer which was used to count Gamma rays in the energy range of 220 to 384 kev. The canisters are counted for thirty minutes in a low background environment of approximately 600 counts. The conversion from these net counts to radon concentrations is determined via calibration factors determined by exposing canisters to known radon 222 concentrations in a radon chamber at East Stroudsburg University. Calibration factors were calculated for two different relative humidities of 10% and 80% and numerous radon 222 concentrations. Linear interpolation between these values is used to determine radon concentrations for intermediate ranges.

A daily quality assurance check on the counting system was performed with canisters which had been exposed at known concentrations. Further calibration results were made by an external lab.

Distribution of Radon Test Kits

Results reported here have been based on the kits which were returned to the University from January, 1986 to April, 1989. This is an actual case study, not a random sample study.

Included in the kits is a one-page list of instructions for the proper use of the canisters. These instructions are in accordance with EPA protocols.

A brief questionnaire to collect data on home characteristics is also included with the radon test kit. The earlier ones that were handed out from January, 1986 to January, 1987 consisted of a questionnaire that asked for the zip code, county, the location of the detectors within the house, the number of stories, the relative humidity, and the type of heating system used in the home. Questions were asked, such as: Does the house have a crawl space, concrete slab, or basement? (This is unlike the new questionnaire presently in use, which is much more elaborate, having twenty-nine questions.)

Discussion

Figure #1, shows a map of radon concentrations with shading from the lightest to darkest, indicating radon concentrations from low to high concentrations respectively. This figure also shows that there are six zip codes in Monroe County whose mean radon concentrations are lower than 4 pCi/L, as shown in Table 1. The highest mean concentration is in zip code 18341 with a mean concentration of 49.2 pCi/L.

Figure #2 shows Monroe County again by zip codes, the white areas representing those regions where no measurements exceed 10 pCi/L. The area represented by the hash marks shows regions where no measurements exceed 41 pCi/L, and the black represents areas where no measurements exceed 420 pCi/L.

Table 1 depicts the percentage of homes whose radon concentrations are below the value indicated.

Figure 3 compares radon concentrations in homes with well water to radon concentrations in homes with municipal water supplies.

In Monroe County there is approximately an equal number of homes utilizing municipal water supplies as there are using well water. This data is heavily weighed to those homes having municipal water supplies. A further categorization of the data is by floor level --basement, first floor, and second floor--also shown in Figure 3.

On all three floors (basement, first, and second), there was no significant difference between radon concentrations in homes which use well water and those which use municipal water. In fact, homes which used municipal water had a higher mean radon concentration on all floors compared to the radon concentrations found in homes which used well water.

Conclusions

It should be noted in Figure #1 that the darker colored lines are in the southern part of Monroe County, which is close to the Reading Prong, yielding one possible reason for the southern end of Monroe County to have such a high radon concentration. It could also be speculated that when the glaciers moved southward towards this area during the last ice age, they advanced not too much farther than the southern end of Monroe County. When these glaciers receded they uncovered the top layer of soil leaving uranium deposits close to the surface.

It is of some interest to note that 42.4% of the homes tested in Monroe County have radon concentrations greater than 4.0 pCi/L. This is in modest agreement with the most recent Pennsylvania D.E.R. results, whereas early radon concentration estimates by EPA indicated 15% of all U.S homes would have radon concentrations greater than 4.0 pCi/L.

In his book, Radon: A Homeowner's Guide to Detection and Control, Cohen indicates that of 6606 homes he tested in Pennsylvania, 3055 homes (46%) were above 4 pCi/L. Michael Lafavore points out in his book, Radon: The Invisible Threat, that of 20,527 homes tested in Pennsylvania, 11,770 were above 4 pCi/L or 57.3% of homes tested.

If half of all homes in Monroe County were to be mitigated at an average cost of $1,500.00/home (this estimate allows for sub-slab ventilation), the total cost would become 67.5 million dollars. Too, one might note that original EPA estimates indicated only 15% of all homes in the U.S. might need to be mitigated, to be lowered to below 4.0 pCi/L.

Another parameter of interest is the number of measurements per square mile. Pennsylvania covers an area of approximately 45,333 square miles. The 6606 measurements obtained by Cohen, yield a density of .15 measurements per square mile. Lafavore's measurement's yield a density of .45 measurements per square mile. This indicates that even though these measurement densities are less than that of this report the results are all in very close agreement.

The results in Figure 3 are somewhat surprising in that one would expect that homes with well water would have higher radon concentration than those with municipal water supplies (even though the data in not significantly different). Two possible explanations are proposed:

1. There are many small municipalities with their own water supply coming from small reservoirs that are replenished by a stream.

2. A number of municipalities obtain their water from deep wells.

There are 90,000 full-time homes in Monroe County, and at the present time E.S.U. has tested almost 1% of these homes. The data depicted in Figures 1, 2, and 3, of course, report on all the data. However, if one chooses at random 10% of this data to analyze, then 20%, etc., one finds that results identical to within 20% of those reported herein are produced when only 20% of the data are analyzed. This suggests that one needs only .2% of all homes to be measured to determine average radon concentration values to within 20% of a study analyzing five times more data. A further complication is indicated in these results by the possibility that a given homeowner may have evaluated the radon concentration in a given home more than once (perhaps before and after mitigation). Because of the confidentiality of the homeowner's name and address, and because many of the addresses are rural routes, we can only estimate this effect. We estimate less than 1% of the data is so affected.

One last important note. Figures 1 and 2 are somewhat misleading in the depicting actual boundaries for given average radon concentrations. The boundaries are by zip code and the average radon concentrations are also reported by

zip code. Average radon concentrations taken over the same county but divided geographically in other ways could yield results different in some respects from those reported here.

The work described in this paper was not funded by the U.S. Environmental Protection Agency and therefore the contents do not necessarily reflect the views of the Agency and no official endorsement should be inferred.

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Table 1. Radon Levels in Monroe County by Percentage

Radon Level (L) (pCi/L)	% of Homes with Radon Less than (L)
4	57.6%
10	79.9%
15	86 1%
20	88.9%
25	91 5%
50	96.2%
100	98 8%
420	100.0%

NUMBER OF SAMPLES	MEAN pCi/L	ZIP
10	1.7	18325
3	1.7	18445
4	2.5	18610
33	2.9	18335
12	3.0	18347
2	3.4	18334
7	4.1	18352
21	4.4	18466
31	4.5	18372
24	5.0	18326
1	5.1	18320
5	5.1	18333
3	5.3	18342
16	5.6	18350
1	6.3	18323
13	6.7	18322
180	7.2	18301
5	7.5	18356
18	8.1	18370
8	8.5	18355
12	8.7	18354
205	10.5	18360
71	10.8	18344
12	13.7	18332
9	15.1	18327
4	17.3	18331
2	17.4	18349
22	21.7	18058
47	24.0	18353
5	26.3	18321
26	33.9	18330
4	49.2	18341

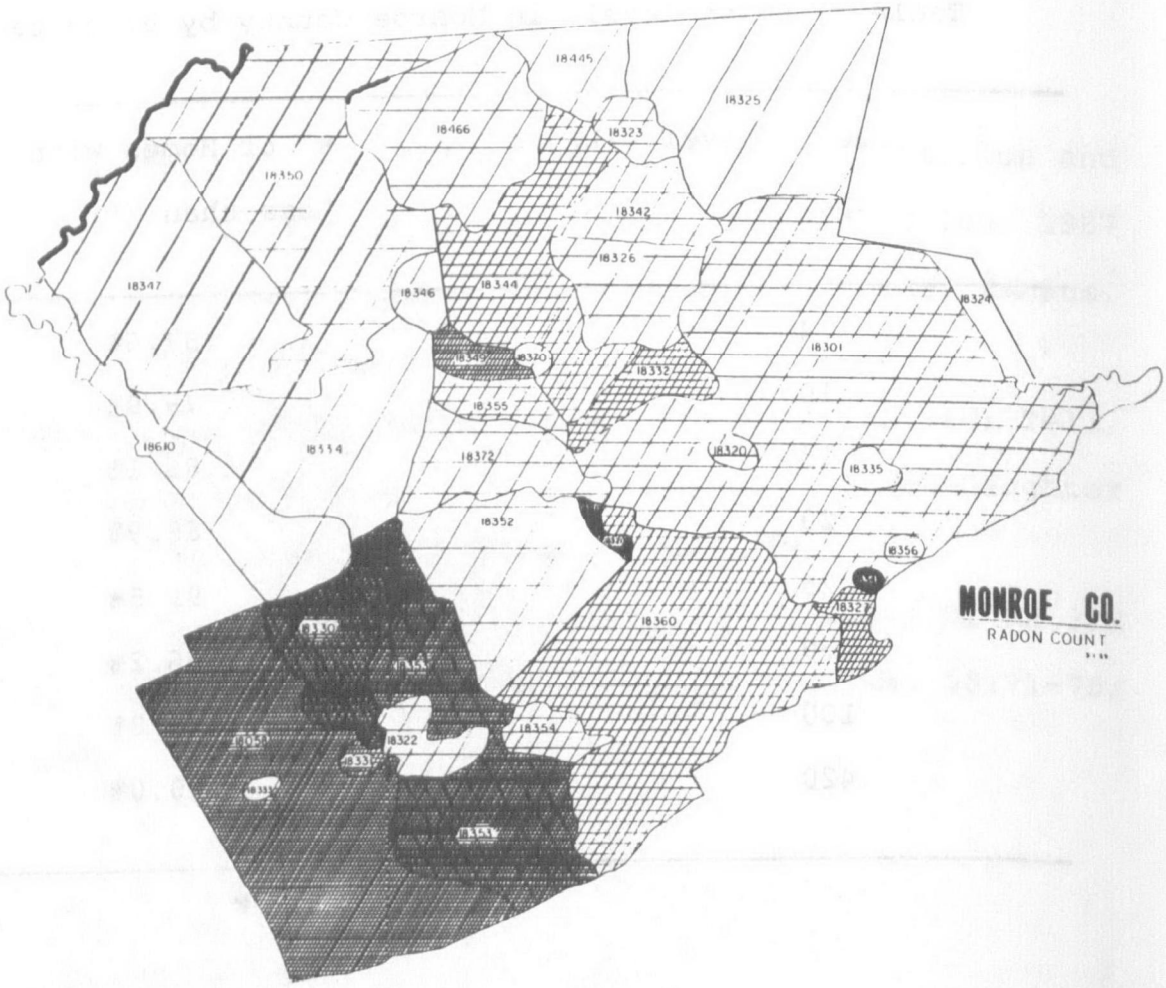

FIG. 1 MAP OF RADON CONCENTRATIONS IN MONROE COUNTY. LIGHTER SHADED AREAS INDICATE LOW CONCENTRATIONS. GEOGRAPHIC DIVISIONS ARE BY ZIP CODE.

P.N. Houle & D.N. Blick

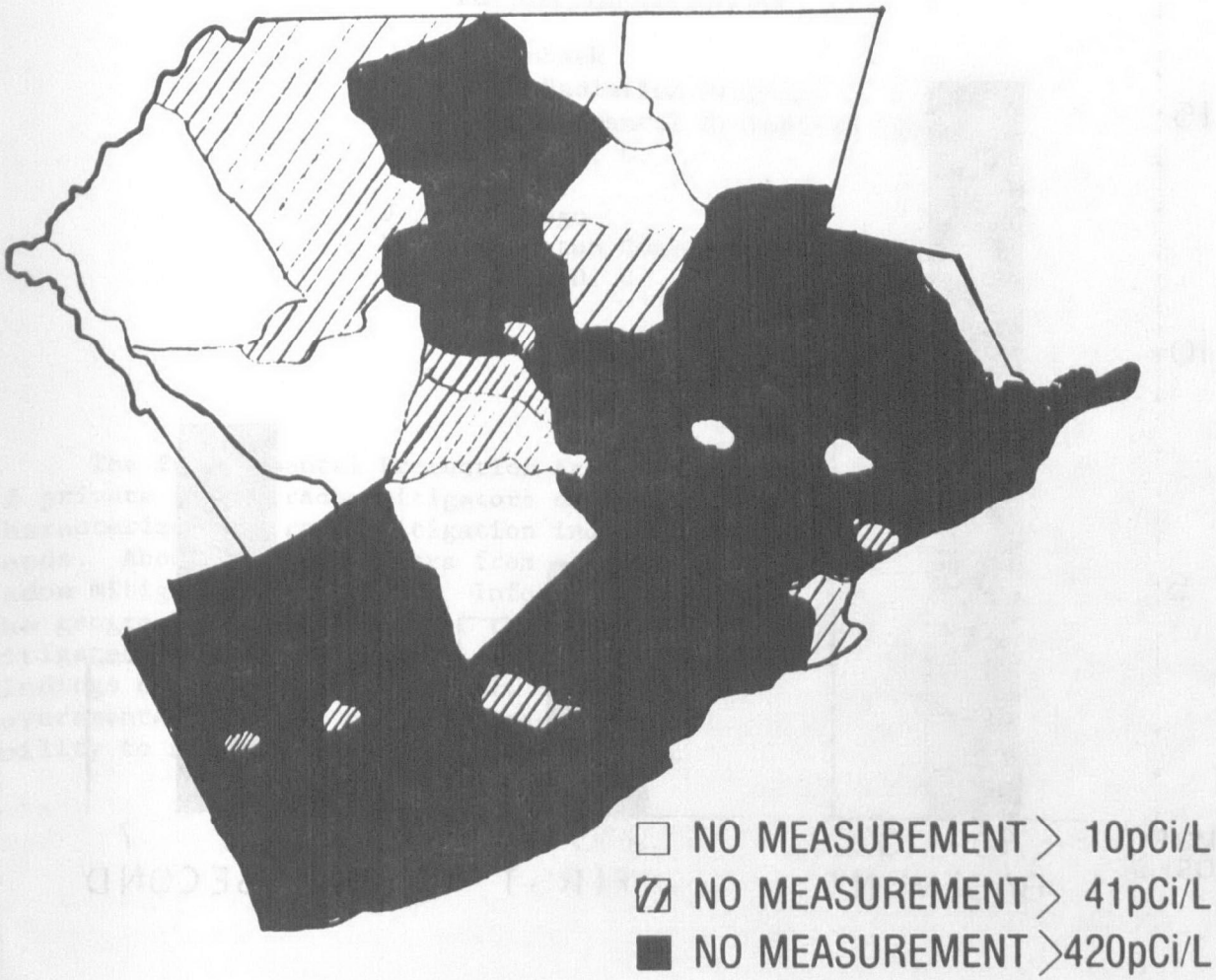

FIG. 2 MAXIMUM Rn^{222} MEASUREMENT IN MONROE CO.

P.N. Houle & D.N. Blick

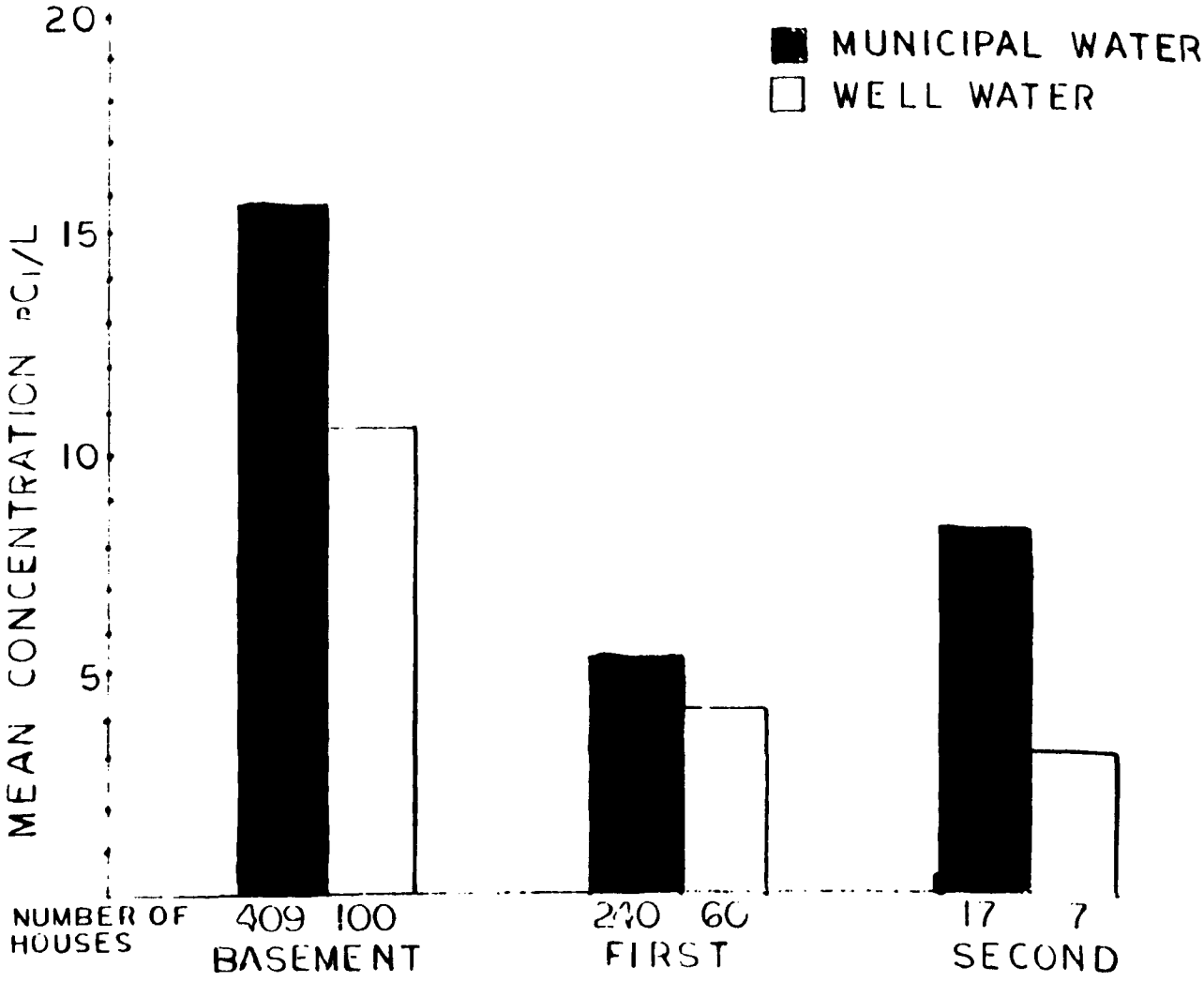

FIG. 3 MEAN Rn^{222} CONCENTRATIONS FOR HOUSES ON MUNICIPAL WATER AND WELL WATER

P.N. Houle & D.N. Blick

PRIVATE SECTOR SURVEY

by: John Hoornbeek
Office of Radiation Program
U. S. Environmental Protection Agency
Washington, D. C.

Josefina Lago
The Washington Consulting Group
Washington, D. C.

ABSTRACT

The Environmental Protection Agency (EPA) conducted a nationwide survey of private sector radon mitigators during the summer of 1989. The survey characterizes the radon mitigation industry, its accomplishments, and its needs. About 1700 mitigators from all 50 states were questioned on their radon mitigation activities. Information gained through the effort includes the geographic distribution of the industry, the number of buildings mitigated, industry growth rates, and training needs. This paper reviews the findings of the survey in detail. It also provides suggestions on how governmental policies and practices can be changed to improve the industry's ability to reduce the health risks of radon.

SAMPLING STRATEGIES OF RADON SURVEYS: THE ITALIAN EXPERIENCE

by: G.Campos-Venuti and G.Farchi
Istituto Superiore di Sanità, V.le R.Elena 299
00161 Rome, Italy

S.Piermattei
ENEA/DISP, Via V.Brancati 48, 00144 Rome, Italy

ABSTRACT

The criteria according to which these surveys are carried out are of particular importance in obtaining reliable results. The present paper is concerned with the methodologies employed in national surveys of the second generation to evaluate the mean exposure of the population to natural radiation indoors.

The paper describes the probability sampling procedure (a two-stage stratified technique) adopted in Italy to carry out the national survey. This type of sampling (with 5000 dwellings) will allow to obtain representative results both at Regional level and National level. This is particularly relevant since the survey represents a cooperative effort between national and local (administrative and scientific) organizations.

INTRODUCTION

An extensive research work (1, 2, 3) has been conducted during the last twenty years by a great number of laboratories. The aim of this work consisted in the evaluation of the amount of the exposure to natural radiation in the domestic environment mainly with regard to radon and its decay products and in the investigation of the various parameters affecting their presence indoors (i.e., geology of soil, microclimate, ventilation, building materials, water supply, heating systems, etc).

Several countries undertook surveys in dwellings (see, for instance 4 and 5), called first generation surveys, to determine the ranges of indoor radon concentration and to identify high risk areas. Theoretical and experimental studies were also carried out to improve the knowledge of the physical behaviour of radon decay products (aerosol size distribution,

diffusivity, attachment rate, etc.) in the room air, to measure and model aerosol deposition to better understand the role of the unattached fraction of aerosols in dose calculation (6,7).

Towards the middle of the eighties the knowledge of radon entry routes was improved and was evidentiated the major role played by the soil as source of indoor radon compared to building materials, not only with regards to the Ra-226 content but also to other important soil parameters such as grain size, permeability, porosity, moisture content, etc. (8). At this time, studies were also carried out which demonstrated the existence of an inverse relationship between the attached fraction of radon daughters existing in the domestic environment and the degree of equilibrium of radon gas and its progeny with respect to the aerosol concentration. This suggested a direct relationship between the dose and the concentration of radon gas, which can be more easily measured by integrating passive detectors (9). During these years reviews of epidemiological studies on underground miners, together with the improvement of the existing dosimetric models, allowed the validation and refining of the risk estimates associated to exposure to radon indoors (10,11).

ICRP in its Publication 39 (12) suggested intervention levels at which remedial actions might be undertaken in buildings where radon concentration exceeds such values and, at the same time, a distinction was made between old and new dwellings. Some countries (USA, U.K., Nordic countries) have adopted recommendations to limit the exposure to radon daughters indoors by choosing Action Levels for existing and upper bounds for future houses. For these reasons new series of surveys, called <u>second generation</u> surveys, were undertaken with two main objectives:

1) the evaluation of the average health risk to the population deriving from exposure to indoor radon;
2) the knowledge of the number of dwellings where proposed levels are being exceeded and the assessment of economical and social costs of remedial action nationwide.

These objectives can be attained only provided that the measurements are carried out by selecting a sample of dwellings statistically representative of the national situation.

This paper describes the various sampling strategies employed to choose a representative sample of dwellings and illustrate the one adopted in Italy for the National survey to evaluate the exposure of Italian population to natural radiation indoors.

SAMPLING STRATEGIES

The basic concept to be kept in mind when a sample survey is undertaken is that reliable information will be obtained provided that: i) the sample is <u>representative</u>, i.e., reflects correctly the units under study and it is possible to draw an inference from the sample to the population; ii) the sample is large enough to yield significant results.

Different methods are available to select a representative sample (13). The most straightforward one is called List sampling, or Random List sampling. Every sample has the same chance to be selected and the application of the method is subjected to the availability of a reliable and up-to-date list.

The Stratified sampling method can be used when similar items have to be studied and comparisons are to be made among them. In this context, stratum means a group of study units either rather similar or placed in similar situations. The strata may differ from one another, but within each stratum the study units can be considered similar. In stratified sampling the items under study are divided into a given number of strata and a random selection of study units is drawn from each stratum. This obviously implies the existence of a list of study units. The stratified sampling technique offers two main advantages: i) for the same sample size the information obtained are more reliable if the items to be surveyed can be divided into strata and ii) comparisons among them can be made.

When the list of study units is not available and its preparation would be too expensive, the cluster sampling method can be applied. In this case the sampling units consist of a cluster of smaller items, close together, obtained, for instance, by dividing the survey areas into smaller ones, where it is possible to make a list of the study units. The size of the cluster, as a general rule, should be small.

In some instances it would be more convenient to use a multi-stage sampling procedure with separate sampling at each stage.

The sample size should be large enough to give reliable results. The ideal sample would be formed by all the units under study. This approach being generally impracticable, the size of the sample should represent a compromise between the need of reliable results and the availability of economic resources and manpower. This decision is based on two considerations: 1) on the basis of the resources the number of study units to be processed is established; 2) the reliability offered by the chosen sample size is evaluated by using either previous measurements or results from pilot studies.

SAMPLE SELECTION METHOD USED IN THE SECOND GENERATION ITALIAN NATIONAL SURVEY

From the above discussion, it appears that the choice of the sampling method is strictly related to the existing information and to their reliability. When the decision was taken to carry out the national survey on natural radiation indoors, two distinct categories of information were available on the sources and the places of exposure (dwellings).

The first one includes the geology of soils; the measurements of absorbed dose rate in air outdoors covering most of the country (measurements carried out in the early seventies); the radioactivity content in typical building materials and in mineral waters from particular springs; the radon content in the deep soil of some areas; the radon concentration

and gamma ray absorbed dose rate in dwellings (14, 15). The characteristics and the use of these information will be discussed below.

The second category includes data on the dwellings obtained through the general population Census carried out in 1981 (16). These data concern the following:

i) type of house: isolated family dwelling or family dwelling part of a development;
ii) characteristics of building materials: concrete or other;
iii) number of apartments in the same building;
iv) period of construction: before 1919, 1919-45, 1946-60, 1961-71, 1972-75, 1976-80, after 1980;
v) source of drinkable water: aqueduct, well, other;
vi) heating system: central or other, type of fuel.

The final Census data are published and available to the general public. The Census data include also the number of inhabitants and families in each Unit (Comuni) and District (Regioni). These figures are updated yearly and stored in a file to which our organizations have access.

The choice of a list sampling method was excluded a priori since a list of all the dwellings is not available. The use of telephone directories was also excluded since the telephone density files - where telephone density is defined as the ratio between the number of telephones and the number of dwellings in each District - prepared by the National Telephon Company showed large variations from place to place. This would have introduced a strong bias in the sample. Identifiable groups such as health physicists, civil servants, volunteers, cannot be used for a representative survey.

The only option left was a probability stratified sampling cluster technique. The most important point was the selection of the strata. Given the paramount role attributed to the soil in determining indoor radon concentration, the results of the geological surveys carried out during the sixties were analysed. A cooperative effort involving the Department of Earth Science of the University of Rome, ENEA/DISP and the Istituto Superiore di Sanità (17) resulted in the partition of the country into 6 geological groupings (Figure 1) (recent sediments, argillaceous flysch, limestones, igneous metamorphic rocks, volcanic basic rocks, recent volcanic rocks) as strata for sample selection purposes.

Using information derived from the Census a further partition could be envisioned considering the construction period of dwellings which corresponds to different construction styles either for the use of building materials or for architectural design (the seven periods of construction in the Census correspond to important steps in the evolution of building typologies). The partition into geographical areas corresponding to significant differences in climate affecting the indoor radon concentration could be used.

The types of stratification examined, although valid from the scientific point of view, showed some drawbacks. The administrative borders

were not respected and the access to the Census information proved rather difficult. In fact, during the Census data processing the addresses of the dwellings were not recorded and the link between the data on the house and its address is practically impossible. Moreover, analysis of manpower and economical resources evidentiated that no more than 5000 dwellings located in 200 Administrative Units could be included in the survey. Being this the case, the use of a too large number of strata would have diminished the significance of the information collected.

Furthermore different needs arose during the discussions of the sampling strategy. First, the radon concentration indoors represents mainly a health problem and, in this sense, the two leading Institutes considered important the involvement of local health Authorities and gave them the opportunity of contributing to the survey. Consequently, the sampling

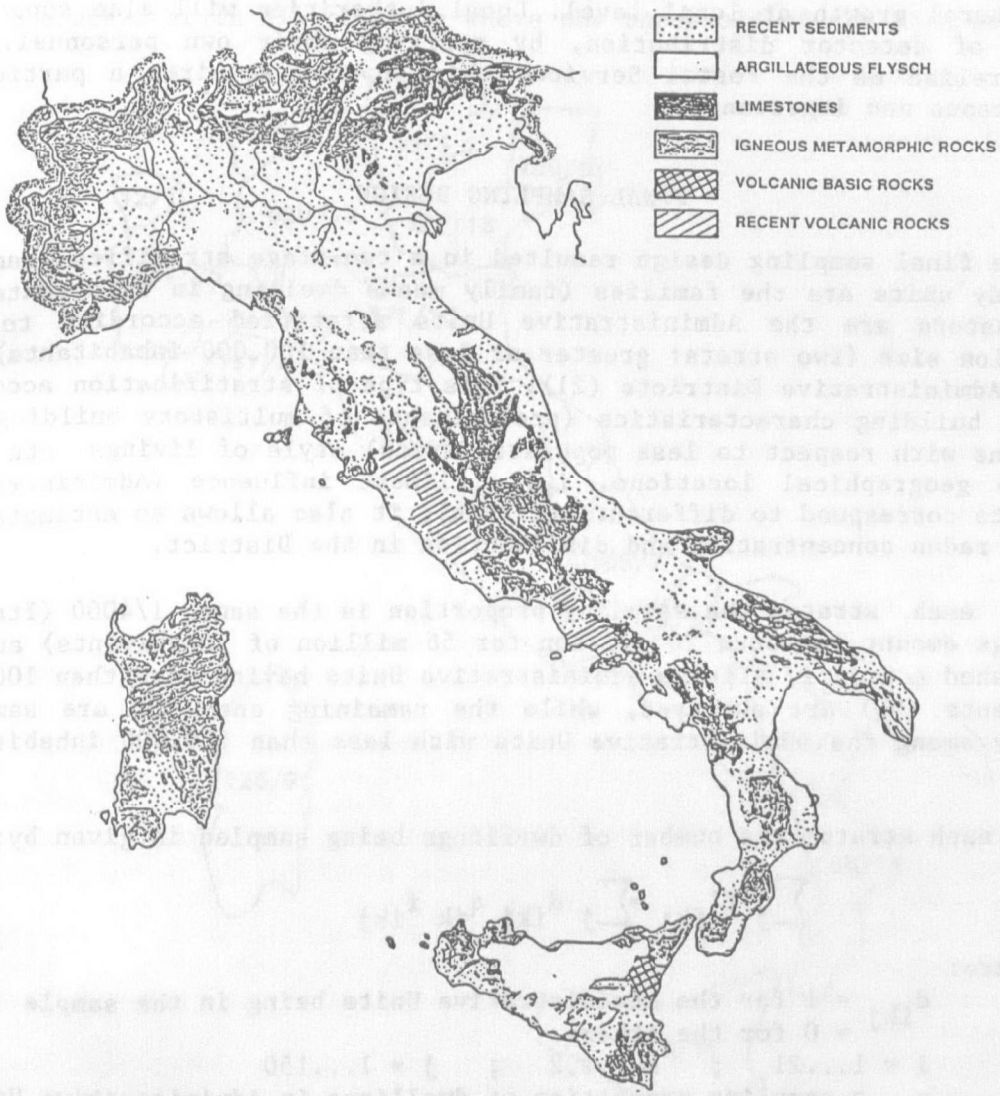

Figure 1. Partition of the Italian territory into 6 geological areas.

strategy should provide a sample of dwellings statistically representative of the situations existing in the Administrative District. Secondly, the knowledge of radon distribution in towns where the building criteria show large differences (urban and less densely populated areas) is of particular relevance.

According to the first one of the above considerations, it should be remembered that in Italy in 1978 the National Health Service was established by law and that activities involving health matters are delegated to the Administrative Districts by the State. The control of environmental radioactivity is one of these activities. In 1987 the Ministry of Health issued a Directive to the Administrative Districts concerning such control and laboratories are being set up for this purpose. In this context the creation of research units dedicated to radon measurements was considered of paramount importance, as a valid contribution to the cultural growth at local level. Local Authorities will also support a program of detector distribution, by applying their own personnel. Not having relied on the Postal Service, this support acquires a particular significance and importance.

FINAL SAMPLING DESIGN

The final sampling design resulted in a two-stage stratified sample. The study units are the families (family means dwelling in this context); the clusters are the Administrative Units stratified according to the population size (two strata: greater or less than 100.000 inhabitants) and to the Administrative Districts (21). This type of stratification accounts for the building characteristics (predominance of multistory buildings in big towns with respect to less populated areas) style of livings, etc. and for the geographical locations, i.e., climate influence (Administrative Districts correspond to different latitude); it also allows to estimate the average radon concentration and distribution in the District.

In each stratum the sampling proportion is the same: 1/4000 (Italian dwellings amount to about 20 million for 56 million of inhabitants) and is established a priori. All the Administrative Units having more than 100.000 inhabitants (50) are surveyed, while the remaining ones 150 are sampled randomly among the Administrative Units with less than 100.000 inhabitants (8042).

In each stratum the number of dwellings being sampled is given by:

$$\sum_j f'_{ikj} = \sum_j d_{ikj}\, q_{ik}\, f_{ikj}$$

where:

d_{ikj} = 1 for the Administrative Units being in the sample
= 0 for the others

$i = 1...21$; $k = 1,2$; $j = 1...150$

q_{ik} = sampling proportion of dwellings in Administrative Units of stratum ik =

$$= \frac{P \sum_j f_{ikj}}{\sum_j d_{ikj} f_{ikj}}$$

f_{ikj} = number of dwellings in Administrative Unit j of stratum ik

f'_{ikj} = number of sampled dwellings in the Administrative Unit j of stratum ik

P = final sampling proportion

The clusters are sampled at central level using the updated Census files, the dwellings are sampled at local level where the family registers are available. On the basis of the measurements carried out in previous years in Italian dwellings, the sample size chosen (5000 dwellings distributed in 200 Administrative Units) will allow to obtain an indoor radon concentration distribution where the threshold of the 95th percentile

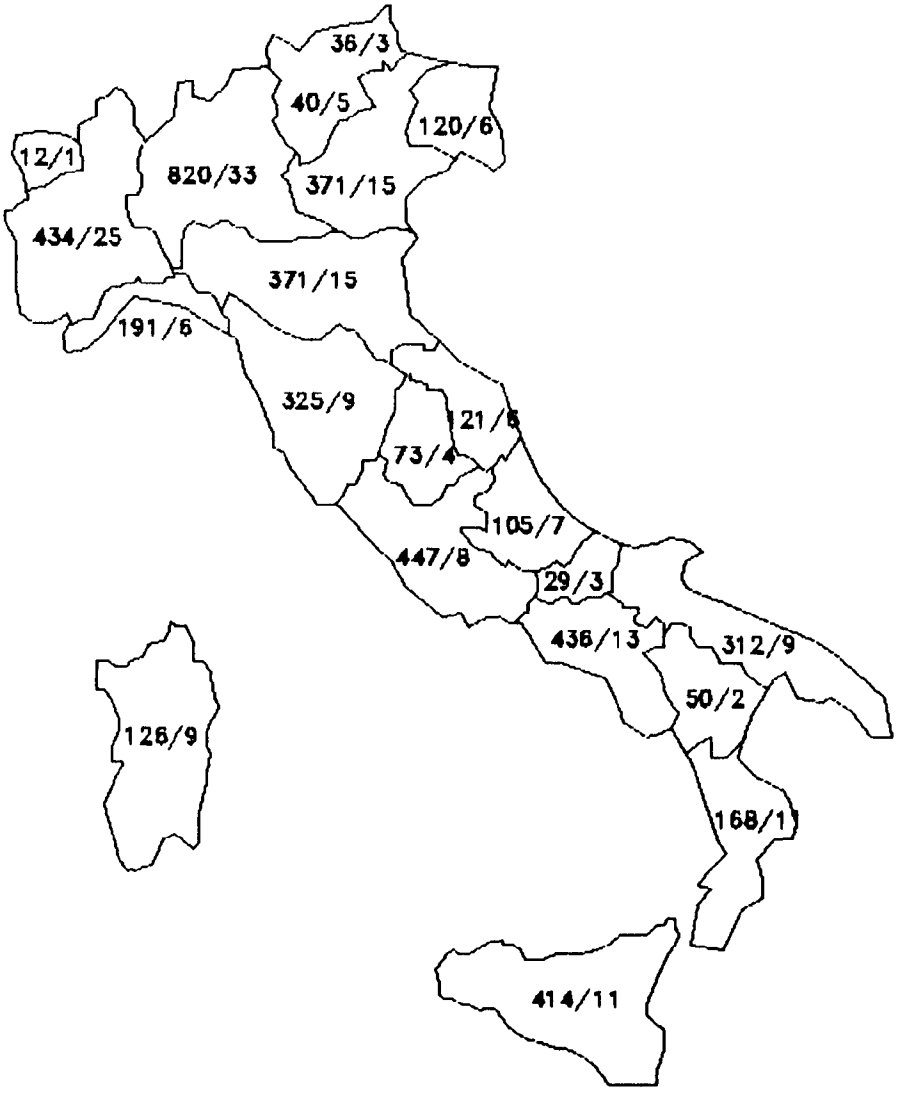

Figure 2. Number of dwellings/Administrative Units in each District.

will be evaluated with a standard error less than 10%. For an average Administrative District with 250 dwellings the same threshold will be affected by a standard error around 25%. The distribution of the dwellings in the sample is shown in Figure 2.

The survey is supplemented with a questionnaire carefully studied. It contains a set of information which should allow to correlate indoor radon concentration with other relevant parameters. The questionnaire will be filled by personnel in charge of the distribution of the dosemeters, trained for the purpose.

CONCLUSIONS

The efforts made to carry out the sampling represent an example of a cooperative process among central Institutes and local Authorities, which was deemed important for two main reasons: a national survey implies financial and manpower burden for a country and has to be carefully planned in order to get the best information, considering also the social and economical implication which could derive from it; the involvement of the local health Authorities contributes to the creation of a correct expertise in environmental problems.

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FIRST-PHASE STUDY DESIGN FOR THE U. S. NAVY RADON ASSESSMENT AND MITIGATION PROGRAM (NAVRAMP)

R. B. Gammage, D. L. Wilson, C. S. Dudney, and T. G. Matthews
Oak Ridge National Laboratory
Health and Safety Research Division
P. O. Box 2008
Oak Ridge, Tennessee 37831-6383

ABSTRACT

In 1988, the Navy initiated a multi-year program for the assessment and mitigation of radon inside buildings at its worldwide distribution of bases. During the first two years of the program, a survey is being made of indoor radon levels in residences occupied by Navy personnel and their dependents. In addition, a small random sample of other structures is being monitored for elevated radon. Passive alpha-track detectors, numbering about 25,000, are being used as monitoring devices. A substantial fraction of the monitors (20%) are being used for quality assurance. Data management programs have been developed to record the chain of custody of the monitors and handle the associated questionnaire data. Program objectives and implementation emphasize quality assurance, records maintenance and monitor placement and retrieval.

INTRODUCTION

During the last decade there has been a gradual realization that indoor concentrations of radon pose a significant national and international problem for radiation exposure to the lung. For example, Nero et al. (1) estimated that about one million U.S. homes have radon levels exceeding 8 pCi L^{-1}. The U.S. Environmental Protection Agency (EPA) is currently recommending (2) that all residences be screened for radon. The recent Indoor Radon Abatement Act (3) declares the national goal is that "air within buildings in the United States should be as free of radon as the ambient air outside of buildings."

The EPA recommends that residences be screened for radon and that 4 pCi L^{-1} be the level above which remedial action should be considered (4). This action is predicated more by pragmatic experience in successfully mitigating houses than it is by health risk considerations. The first-phase screening portion of the U.S. Navy Radon Assessment and Mitigation Program (NAVRAMP) project aims to identify with reasonable certainty those bases at which one or more buildings have a time-integrated indoor radon concentration of 4 pCi L^{-1} or more.

The Department of the Navy (DON) has 100,000 structures under its jurisdiction, ranging from detached, single-family houses to large hospitals. To investigate radon levels in every structure in an acceptable manner would be a very time consuming and expensive, if not impossible, task. In lieu of investigating every structure, DON established the NAVRAMP project which includes an initial phase to establish priorities at U.S. naval facilities for additional radon assessment and possible mitigation. This report will describe the measurement technique and sampling strategy used for the prioritization phase. In addition, it will discuss preliminary plans for analyzing and reporting data.

MEASUREMENT TECHNIQUES

ALPHA TRACK DETECTOR (ATD)

The naval bases being examined include remote and inaccessible sites. Mail service is often non-routine. The charcoal canister (CC) method for radon screening is quite unsuitable for use in such situations; it is necessary to effect prompt return of exposed charcoal canisters back to a laboratory for counting in order to obtain reliable results. The plastic, alpha-track recording detector, in contrast, provides a nearly non-fading record of any radon exposure. Delay in returning an exposed ATD to the laboratory for development has little effect on the result. The ATD was thus chosen for the NAVRAMP project, even though the ATD is about twice as expensive as the CC.

VENDOR SELECTION

The selection of a commercial vendor to supply the ATDs and provide readout service was made using competitive procurement procedures. Specifications describing NAVRAMP requirements were prepared and distributed to all known U.S. vendors having proficiency certification. These were vendors currently listed as having passed the most recent round of EPA's

Radon Measurement Proficiency Program. The vendor providing the lowest bid was inspected by our Program's Quality Assurance (QA) specialist to verify that the vendor's internal procedures were acceptable.

SAMPLING STRATEGY FOR FAMILY HOUSING

EXPOSURE TIME

The EPA recommends screening at a time of the year when the building can be closed up as much as possible; the intent is to ensure that measurement is made at the time of highest and most stable radon concentration (4). It is also recommended that the ATD be exposed for between one and three months so that accurate measurement can be made at concentrations of a few pCi L^{-1}. For an integrated exposure of 40 pCi L^{-1} days, the coefficient variation for a commercial ATD is typically ±25%.

It was necessary to determine which period of the year was the one most appropriate for making a measurement at a particular naval base. Climatic conditions vary in extreme from cool or cold for the whole year (e.g., Adak, Alaska) to tropical (Guam or Subic Bay, Philippines). At other locations the heating or cooling season may be very short (e.g., Key West, Florida). To spread the work load evenly and mesh with the U.S. Navy timetable for completing the screening phase, the following course of action was adopted: (1) exposures would be made for one year at Bases which are in cool-climate zones and for which the buildings remain closed for all or most of the year; (2) in warm or tropical zones, exposures would be made for at least three months when the weather is the hottest and buildings are closed and air conditioned. The first screening results for some Bases with tropical climates will be forthcoming in late 1989.

SELECTION OF SAMPLING SITES

Worldwide, the Navy's inventory shows approximately 95,000 family housing units and 55,000 non-housing structures. The selection of sampling sites at individual bases was a non-trivial task. The approach used for family housing was to obtain a copy of the Navy's data base, update it, and obtain copies of maps of family housing on each Base. We then prepared lists for all Bases describing which structures required detectors. Ninety percent of the detectors are devoted to family housing. The greatest concerns for exposure are in family units where occupancy times can be high and children are often involved.

DATA BASE UPDATE

For the prioritization phase of NAVRAMP, the fundamental group of statistical units for sampling was the family housing area. A housing area is a group of houses, town houses, or apartments, of similar structure, built around the same time, and located close together. Using housing-area data bases, the DON Facility Category Code data base, and maps of the facilities, selections were made based upon the following factors: (1) size of the housing area, (2) structural types, and (3) location of the area relative to other housing areas. The available sources of information were cross-correlated and anomalies were resolved by contacting Base housing officers (or their staff). The information was used to update a data base of housing area information. In

all cases, the ATDs at a given base were distributed in as wide a geographical area as possible. After the selections were made, buildings to receive a detector were identified and marked on the maps. Lists were prepared of buildings in each housing area to receive detectors. If fewer than 33 living units were located in a housing area, every unit was selected to receive an ATD. Otherwise, enough buildings were selected so that detectors could be placed in 33 units. Discretion to reassign a detector to an alternate site was given to Base personnel who actually placed the detectors, in the event that the initial assignment was in error or unforeseen difficulties arose with a given building.

SAMPLING STRATEGY FOR NON-HOUSING STRUCTURES

During the planning stages of the program we discovered that the central Navy data base for non-housing structures was not as current or complete as the program required. To avoid delays in the implementation of the program, we performed a limited evaluation of radon in (1) hospitals, (2) brigs, (3) child care centers, (4) schools, and (5) bachelors' quarters. The main purposes of these evaluations were to familiarize Base personnel with the program, obtain additional information on structural and mechanical systems, and obtain an accurate inventory of non-housing structures. Detectors were allocated, either on a per-Base or a per-building basis, as shown in Table 1. A more detailed evaluation of radon in non-housing structures will be performed in later phases of the NAVRAMP program.

Table 1. Allocation of detectors to non-housing structures.

	Number of detectors per building	Number of detectors per facility
Hospital	----------	3
Brig	----------	2
Child Care Center	2	----------
School	2	----------
Bachelor Quarters	----------	4

DATA QUALITY ASSURANCE

In any large survey, quality control is essential. EPA has prepared a report describing suggested protocols for radon surveys (5). Table 2 summarizes the recommendations from that report. For this study, the EPA recommendations were followed as closely as was practical. For each housing area, two detectors or 5% of the detectors (whichever was greater) were allocated for field blanks. Three detectors (or 10%) for each housing area were designated for replicate (i.e., colocated) measurements. From every manufacturing lot (~660 detectors), 33 detectors were set aside for laboratory spiked

Table 2. EPA recommendations for data quality assurance.

Field Blanks	5%
Colocated Detectors	10%
Positive Controls	a few%
Negative Controls (Lab Blanks)	a few%

samples and blanks. Eight detectors (of each 33) were held as laboratory blanks. Two of these were kept in the original envelope, the rest were opened and kept with the unopened detectors in a radon-free, environmentally controlled environment. For two months, fourteen detectors (12 opened, 2 closed) were exposed at 6 pCi L^{-1} of radon in an environmentally controlled chamber. During the exposure, two detectors were kept in the original envelope. For two months, five detectors (three opened, two closed) were exposed to 100 pCi L^{-1} in an environmental chamber. The other six detectors from each lot were held in reserve in case of mishap or for confirmatory purposes.

In addition to the quality control samples handled by laboratory personnel, there were quality control samples handled by field personnel. At about 10% of the indicated sampling sites, field personnel were instructed to deploy duplicate, i.e., colocated detectors. Travel-type control detectors were shipped to each facility and will be returned without ever having been opened. Exposure-type control detectors were shipped to each facility and will be stored on site until just prior to the return shipment. At that time, the detectors will be sequentially removed from their foil pouches, handled as if they were to be deployed, repackaged, and returned.

Verification of chain of custody of detectors and related data are also very important. A shipment tracking system was developed to record when detectors were shipped, when they were received, when a custody report from the point of contact was received, and when placement data sheets were received. A data form shipped with each detector was to be filled out when the detector was placed at a designated sampling site. These forms record information about monitoring location and time. The placement data forms will be returned and entered into a computer data base. One month before desired retrieval times, notices will be sent to facilities, notifying the points of contact when to begin retrieval. Shortly before the desired time of retrieval, a retrieval data form will be generated for each detector and forwarded to the responsible facility. Data from placement and retrieval will be compared to reduce the incidence of misrecorded information on detector placement.

Instruction packets were prepared to guide Navy personnel in the proper placement of the detectors in: (1) family housing units, (2) bachelors' quarters (officer and enlisted), (3) schools, (4) child care centers, (5) hospitals, and (6) brigs. These packets will be revised and reissued for subsequent phases of NAVRAMP.

IMPLEMENTATION

POINTS OF CONTACT

Implementation of the above sampling design relied on extensive cooperation from Navy personnel, both civilian and military, at each facility. Early in the programs, each Base provided us with points of contact for family housing and for operational structures.

DISTRIBUTION OF DETECTORS

Each shipment of detectors for placement in family housing included the requisite number of detectors, a chain of custody letter/receipt, maps indicating the locations of structures selected for detector placement, instruction packets, and a placement data form for each detector. Each shipment of detectors for placement in non-housing structures was divided into subshipments according to the kind of structure(s); this subdivision was necessary because there were different points of contact for hospitals, brigs, bachelor quarters, schools, and child care centers. Each subshipment included a chain of custody letter/receipt, the appropriate instruction sheet and a detector placement data sheet. In addition, there was a list, derived from our inventory data base, of structures of the appropriate type at that facility and there was a request for building floor plans and additional.

HOTLINE INQUIRIES

To facilitate implementation of this plan by field personnel, a hotline phone number was established to answer questions from the field. The phone was manned during business hours, Eastern Standard Time. At other times, an answering machine recorded messages received, and responses were generated as soon as possible.

DATA MANAGEMENT

To date, over 22,000 detectors have been shipped to naval facilities worldwide. For each of these detectors, information is being compiled on the characteristics of the structure where the detector was placed. Temporal information is compiled on when the detector was shipped to the facility, received, placed, retrieved, shipped back to us, received by us, and shipped to the vendor. The radon exposure (units of pCi L^{-1} d) recorded by the detector is also included. All paper records, including maps, chain-of-custody letters, and detector placement data sheets, are being stored in locked filing cabinets kept in well air-conditioned space at Oak Ridge National Laboratory. To facilitate data entry into computer data bases and assignment of all received information to the correct monitoring site and time, bar coded strips are being used extensively. Bar coded strips are applied to radon detectors and to detector placement sheets. Bar code readers determine detector identity information from the sheets to reduce the occurrence of transcriptional errors. Data entry programs have been developed which compare newly entered data with data already on hand to ensure that only valid identification numbers for radon detectors and Navy buildings are entered.

DATA ANALYSIS AND REPORTING

An integral part of the development of a work plan for NAVRAMP was consideration of how the data would be analyzed.

MINIMUM SAMPLE SIZE

From statistical analysis of assumed populations of houses, it was concluded that if 14% of all units in a population have radon concentrations above 4 pCi L^{-1}, then if at least 30 units are sampled and none of the results are found to be above 4 pCi L^{-1}, the probability of there being any house with elevated radon is less than 0.01 (i.e., 1%). Allowing a 10% outage for inadvertent losses, it was decided to sample 33 units in each housing area with 33 or more units.

DISTRIBUTION OF RADON CONCENTRATIONS IN HOUSING AREAS

After the data are collected, it is possible to assess the distribution of the sample and infer limits on the distribution within the entire population. From statistical analysis, we concluded that, if the population is lognormal, it is possible to estimate the worst-case distribution for a given level of confidence. Using this approach for each housing area, we expect to answer the questions:

1. Does the population follow a lognormal distribution?
2. What is the fraction of the population above any given action level?
3. If we assume that the statistics calculated from the data underestimate the true geometric mean and true geometric standard deviation, what is the fraction of the population above the action level?

Based upon the analysis of radon data from housing units, naval facilities will be ranked for priority scheduling for a more detailed screening survey. No final ranking scheme has been approved, but those facilities found to have a substantial probability of one or more units with radon in excess of 20 pCi L^{-1} will probably receive the highest rank.

ANALYSIS OF DATA FROM LARGE STRUCTURES

The understanding of radon behavior in buildings with complex heating, ventilating, and air conditioning (HVAC) systems is very limited. EPA's research in schools has shown that in some buildings, the induced pressure in different rooms in the same building due to the HVAC system can range from +10 Pa to -20 Pa. It was also shown that even small changes in pressure can markedly affect radon ingress in ground floor rooms. Therefore, we recommend that data from each large building be treated on an individual basis.

This is the first of several reports that are planned. After detectors placed in cooling season facilities are returned to the vendor and results are available to us, the data from those facilities will be summarized as described in this section and a report submitted. Electronic data bases will be prepared and provided to the Navy. The data bases will include information on monitoring sites and times, structure characteristics, and radon concentrations.

ACKNOWLEDGEMENTS

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RADON IN WATER AERATION SYSTEM OPERATIONAL PERFORMANCE

by: Bruce L. Lamarre
North East Environmental Products, Inc.
West Lebanon, NH 03784

ABSTRACT

North East Environmental Products, Inc. is a manufacturer of residential scale aeration systems for removal of radon and volatile organic chemicals from private water supplies. This paper is a review of the operational history of residential scale point of entry (POE) radon aeration systems. Emphasis is placed on the difficulties and solutions encountered in actual installations caused by both mechanical difficulties and water quality parameters. A summary of radon reduction efficiency is presented for wells with radon concentrations from 21,000 to 2,600,000 pCi/L. A discussion of customer concerns and attitudes is presented along with other areas for further technical improvement. Training techniques for dealers and installers are also discussed.

An update of the current status of the radon in water industry includes current sales volumes as compared to the potential market and an update on the radon in water MCL standard setting process from an industry perspective.

BACKGROUND

RADON HEALTH EFFECTS

Radon is slightly soluble in water and can enter a home along with the water from a drilled bedrock well. Because radon is a volatile gas it is quite easily removed from the water when it is used for typical household activities such as bathing and washing dishes. Some estimates based on assumptions about water use patterns and house construction details predict that each 10,000 pCi/L concentration of radon in the water supply will translate into an indoor air concentration of 1 pCi/L. EPA studies have apparently confirmed that this approximation is reasonably accurate for an "average home". If a house has a relatively low air exchange rate this concentration will be higher.

According to health experts, it is the alpha particles from the radioactive decay of radon and its daughters that cause the most severe health threat. Alpha particles are relatively large and will only travel a short distance before striking other matter and give up their high energy. Alpha particles will only travel a few centimeters in air and only, at most, a few millimeters in the human body. Therefore, when a person inhales air containing a high concentration of radon, the most likely organ to be effected by the alpha particles is the lung. In addition, two of the decay products of radon are also alpha emitters. Polonium-218 and polonium-214 are daughters of radon. When radon decays, these elements are formed one after another. Since these elements are solids they attach themselves to dust particles in the air and are carried into the lungs with each breath of air where they decay and give off harmful alpha particles.

Ingestion of water containing radon has been thought to be much less of a problem than inhalation. Because alpha particles cannot even pass through a piece of paper it appears to be very unlikely that when radon, or one of its daughters, decays while located in the stomach or intestine, the alpha particle will travel to and strike the lining of the digestive track. It appears to be more likely that the alpha particle will be absorbed by the fluids in the digestive track and dissipate its energy harmlessly. Of course at extremely high radon concentrations the risk of ingestion damage may become significant.

Since radon is soluble in blood it can be transported throughout the body and potentially affect other organs of the body. Studies are being conducted at the present time by several researchers to establish whether or not radon and its daughters do, in fact, present a health hazard by the ingestion route. Two preliminary papers have in fact shown a very strong correlation between radon in water and various types of cancer. The results are both surprising and alarming.

It has been estimated that of the approximately 120,000 lung cancer deaths each year between 5,000 and 20,000 of those cancers were caused by radon gas exposure. This is the second leading cause of lung cancer. The leading cause is, of course, cigarette smoking. Of the lung cancers caused by radon, the EPA Office of Drinking Water has stated that between 1,000 and 1,800 of these cased can be attributed to radon entering homes through their water supplies. This range of cancer deaths is greater than the cancer risk from all the other water supply contaminants combined. Radon contamination of water supplies is certainly a large and serious problem.

AVAILABLE TREATMENT ALTERNATIVES

Water Treatment - Aeration

Water supplies can be treated through the use of aeration or carbon adsorption techniques. Aeration techniques simply allow the radon to volatilize from the water and exhaust it outdoors where it can disperse harmlessly. Adsorption methods collect the radon on activated carbon and allow it to decay in place.

Aeration devices are relatively simple in that through various methods they contact the water with enough air to volatilize the radon. The treated water then can be pumped into the home water system.

There are basically four types of aeration processes that can be used for residential water treatment: spray aeration, packed columns, diffused aeration and a new process called horizontally extended shallow aeration.

In spray aeration, shown in Figure 1, untreated water from the well is sprayed into a tank through a fine mist spray nozzle. The spray nozzle generates a large amount of water surface area from which radon volatilizes. Usually a small air blower is used to pass a small amount of air through the equipment to carry the radon out of the tank and to vent it outside of the home. Typically a simple spray nozzle will remove approximately 50% of the radon from the untreated water. In order to achieve higher removal efficiencies the water must be resprayed and retreated several times. Essentially any treatment efficiency desired can be achieved with this system. The disadvantage of this process is that the water must be repumped several times, (4 to 5 times usually) and that in order to have a ready supply of treated water the holding tank must be quite large(about 100 gallons). Equipment using this process is available under the trade name No-Rad, (patented).

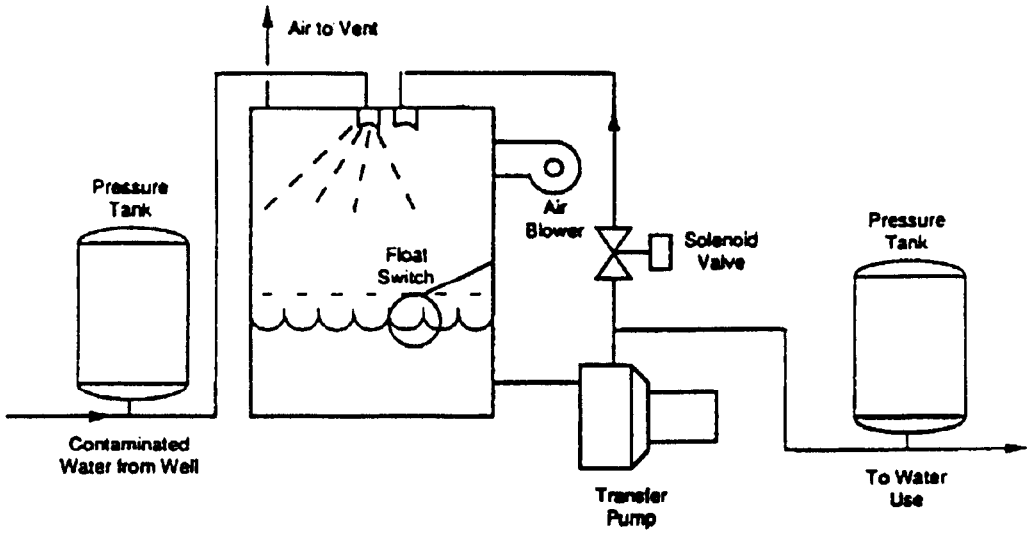

Figure 1. Spray Aeration

Packed columns have been extensively used for removing volatile organic chemicals from contaminated groundwater supplies. These systems can be scaled down for use in a residential setting for removal of very volatile radon gas. Figure 2 shows a residential scale system in which the well water is sprayed into the top of a small air stripping column (available under the trade name Clearadon, patented). The column is filled with about five feet of a common inert dumped packing material. As the water falls down through the packing a large amount of surface area is generated from which the radon can volatilize. A small air blower forces air up through the packing which carries the radon gas out of the column to an outdoor vent. The efficiency of these systems has been shown to be approximately 90 to 95%. The principle limiting factor in packed column aeration of radon is the height available for the air stripping column. Maximum practical packing depth in most residential settings is six feet which produces a removal efficiency of about 95%. For relatively low levels of radon contamination (i.e. up to 20,000 pCi/L) this is entirely adequate. Above this level the packed column system becomes impractical: that is, at a radon MCL of 1000 pCi/L.

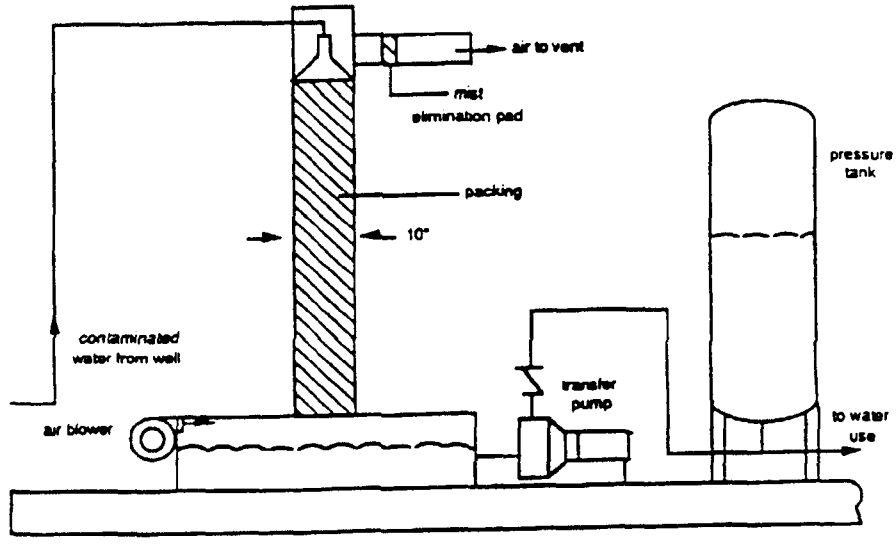

Figure 2. Air Stripper

A third aeration method is diffused aeration. Figure 3 shows this type of process. One supplier manufactures this product (patented) under the trade name of The Stripper. The contaminated well water is pumped into the first of two or more aeration tanks. Air is forced into the bottom of these holding tanks through fine bubble diffusers located near the bottom of each tank by a relatively high pressure (10-15 inches of water column) air blower. As the air bubbles rise up through the water the radon volatilizes into the air bubbles. In this case the mass transfer area for the volatilization of radon from water is generated by the small air bubbles as they rise through the water. Since each tank is essentially completely mixed, very high removal efficiencies cannot be achieved in a single tank. Usually from two to six tanks are required to achieve better than 99% efficiency. The efficiency of an aeration tank can be improved by increasing the residence time of the water in the tank (make the tank bigger), by increasing the number of bubbles (increase the air flow rate and the size of the diffuser) or by baffling the tank so that it performs like a plug flow reactor rather than like a completely mixed reactor.

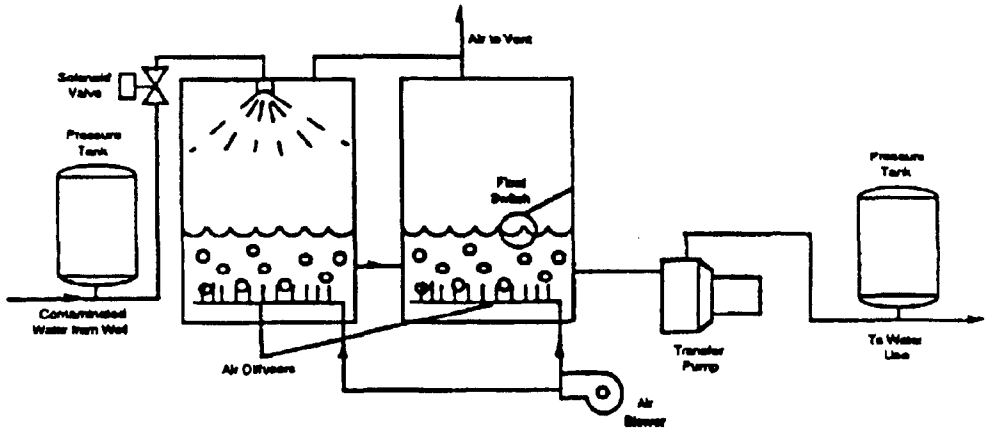

Figure 3. Diffused Aeration

The disadvantages of this system are that a relatively high pressure air blower is required (10 to 15 inches of water column) and that the air holes in the diffuser foul up easily because they are very small, about 0.025 inches.

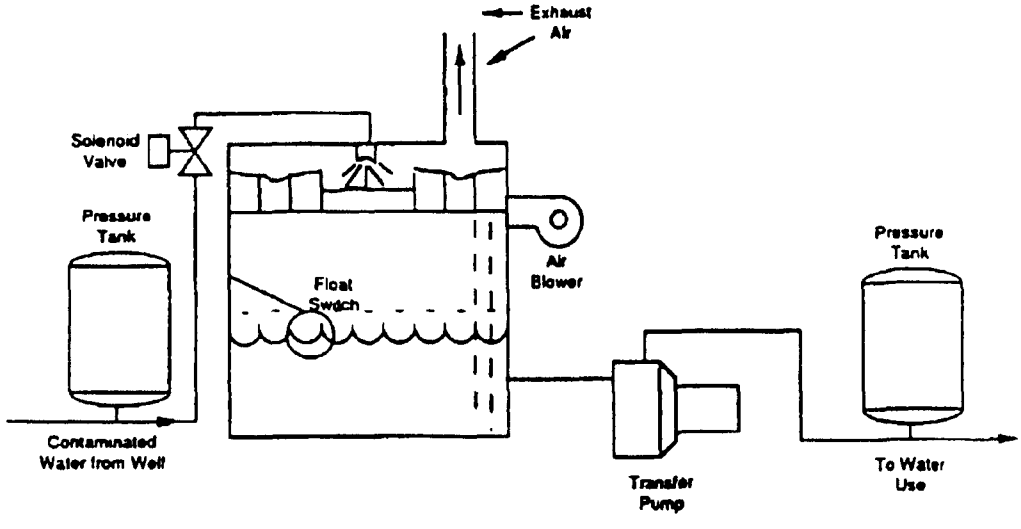

Figure 4. Horizontally Extended Shallow Aeration

Figure 4 shows the forth type of aeration device, the Clearadon 310 radon removal system (patent pending), which uses a horizontally extended shallow aeration tray design to contact the water and air. Water from the drilled well is piped to the aeration unit where it is sprayed into the aeration tray. The water flows across the tray between the baffles as air is blown up through holes in the tray. The air forms a froth of water on the tray creating a very large area for volatilization of the radon from the water. The air evaporates more than 99.9% of the radon, which is vented outside the home. The cleaned water then collects in the bottom of the radon removal unit and is pumped into the water pressure tank. This system, as with the previous systems, is completely automatic and requires very little maintenance. The shallow

aeration system has three principle advantages over the other three systems. First, the aeration tray is smaller than two feet in diameter and only 10 inches high. The overall height of the system can therefore be as little as three feet. The complete system can therefore be smaller, and shorter than the other designs. Second, the air pressure required for operation is only 4 to 5 inches of water column so a much less expensive type of air blower can be used. And third, the air holes in the aeration tray are much larger than those used in the diffused aeration design (3/16 of an inch) and therefore fouling problems are virtually eliminated. The main disadvantage of this system is that it uses approximately 100 cfm of air whereas the previous systems use between 10 and 50 cfm. In some homes an outside air source may be necessary to prevent basement depressurization.

In general, aeration devices have several advantages and beneficial side effects. Aeration devices do not accumulate radioactive elements and therefore do not present a radiation exposure problem and they do not, and will not, need to be licensed as low level radiation sources. Aeration devices can also remove hydrogen sulfide, carbon dioxide, and methane from the water and can assist in the removal of iron and manganese. Aeration devices can also be used to remove volatile organic chemicals such as gasoline components from water supplies.

Water Treatment - Activated Carbon

Adsorption devices remove radon by adsorbing the radon onto the internal surface of a specially prepared "activated carbon". Once adsorbed onto the carbon the radon continues to decay and give off radiation. However, the equipment is usually not located in the immediate living area of the home. After about three weeks, the amount of radon being adsorbed on the carbon equals the decay rate of the radon already adsorbed and the system reaches a steady state. The radiation given off by the unit therefore levels off at some point dependent on the radon level in the ground water.

The advantage of this type of system is that it has very few moving parts and, theoretically, should have quite a long useful life. The disadvantages are the radioactive buildup on the carbon, which may or may not be a real problem depending on the specific situation, the possibility of fouling of the carbon bed, and difficulty in predicting removal efficiencies due to the effects of unknown factors. Contaminants in the water such as iron, manganese, and calcium will be filtered out by the carbon and will eventually plug it. The systems can be cleaned of solids by back washing but this is not completely effective. Activated carbon should not be used for radon treatment of water supplies having over 5,000 pCi/L. The EPA has listed aeration as the best available technology for water system treatment of radon. The liability issue related to the accumulation of radioactive material on the activated carbon alone should be sufficient reason for informed people not to use activated carbon for residential treatment of radon.

OPERATIONAL DIFFICULTIES

There have been three types of operational problems with the shallow tray aeration systems:

- Water leaks
- Noise
- Air in water

As with all types of new mechanical equipment, there have been problems with some of the mechanical components of our earlier systems. The principal problem has been water leaks in the piping system.

WATER LEAKS

Rigid PVC water piping is theoretically, and in the shop, great stuff. It's easy to work with, it has plenty of strength, and it won't corrode. However, threaded PVC piping only has enough mechanical strength if it is not over tightened . It's amazing how easy it is to over tighten a PVC fitting. In about a half dozen cases we have had installations in which a threaded fitting has split shortly after installation. This, of course, caused varying amounts of water damage in some of the homes which we were responsible for correcting. It did not take many of these instances for us to change our design and replace all of the PVC fittings with copper.

Another problem with the PVC is that when it is heated up it looses much of its strength. One of our units was installed at an EPA funded test site in New Hampshire. An unusual combination of settings in the experimental piping set up caused the repressurization pump to run continuously for several days against a closed valve. The water in the pump got hot and the suction piping softened up and swelled. The swelling caused a small leak at a union near the pump.

NOISE

Most residential wells with high concentrations of radon are drilled bedrock wells. A drilled well almost always has a submersible pump mounted in the well to supply water to the house. Submersible wells are virtually silent so home owners are not used to hearing any noise associated with their water supply. Therefore any new noise caused by water treatment equipment may be bothersome to some home owners.

The shallow tray aeration devices have a noise quality which is vary similar to a normal hot air furnace. The noise is not unusually loud but it is unfamiliar in association with the water system. The earlier version of the product had a metal frame that tended to amplify the noise

made by both the repressurization pump and the blower. In order to eliminate this amplification the product was redesigned to completely do away with the metal frame. With out the frame, the pump and blower still generate noise, but at a much lower level.

We also discovered, as I am sure many radon mitigation contractors have also discovered, that rigidly attaching piping to floor joists can also greatly amplify noise generated in pumps and blowers. At a test site installation in Connecticut a very significant noise problem in the living room was eliminated simply by replacing the rigid water pipe clamps with vibration isolation pipe hangers. We later replaced the original Clearadon II with a Clearadon 310 and the noise from the unit was virtually unnoticeable.

AIR IN WATER

In approximately one installation in twenty, we have received complaints from the homeowners that they have air spurting out of their faucet when they turn it on for the first time in several hours. This had been a very difficult problem to correct because of the infrequency and inconsistency of the appearance of the problem.

The cause and solution of the problem turned out to be very simple. Flow restrictors are installed on both the inlet and outlet of the aeration unit. Under some water use conditions and with some well conditions, the amount of water allowed into the aeration unit was slightly less than that being pumped out. This marginal condition allowed a small amount of air to be drawn into the water. The problem was solved simply by installing a slightly larger flow restrictor on the water inlet.

RADON REDUCTION EFFICIENCY SUMMARY

A well located in Dunbarton, N.H. was found to have a radon level of 2.6 million pCi/L. At the time, October of 1988, this was the highest concentration ever found in a private water supply well. Since that time, another well located in Lyons, Colorado has been found to contain 2.9 million pCi/L.

A Clearadon II aeration system was installed at the Dunbarton home. An independent Lab collected and analyzed a set of samples in early February. The results are as follows:

Radon Influent (pCi/L)		Radon Effluent (pCi/L)	
2,565,505		2,773	
2,643,196		1,786	
2,626,649		1,909	
2,468,333		1,754	
2,661,495		1,699	
2,788,235		1,686	
2,625,589	+/- 106.201	1,934	+/- 419

Removal efficiency - 99.93%

The water quality at this home was excellent with the exception of the radon.

Table 1 lists representative performance data for some of the over 100 units we have installed in the field.

TABLE 1. RADON REMOVAL EFFICIENCY

Type of Unit	Inlet Radon Conc., pCi/L	Outlet Radon Conc., pCi/L	% removal	Water Flow Rate, gpm
Clearadon 310	17,000	130	99.24	8.5
	500,000	1500	99.70	8.5
	251,400	540	99.79	7
	172,450	380	99.78	7
	125,200	300	99.76	7
	220,140	210	99.90	5
	141,870	200	99.86	5
	135,900	220	99.84	5
	206,900	40	99.98	3
	165,950	75	99.96	3
	156,800	40	99.98	3
Clearadon II	50,000	>100	99.8	6
	32,000	110	99.7	6
	21,000	>100	99.5	6
	90,000	2100	97.67	6
	57,000	510	99.1	6
	21,000	>100	99.5	6
	2,600,000	1500	99.94	5

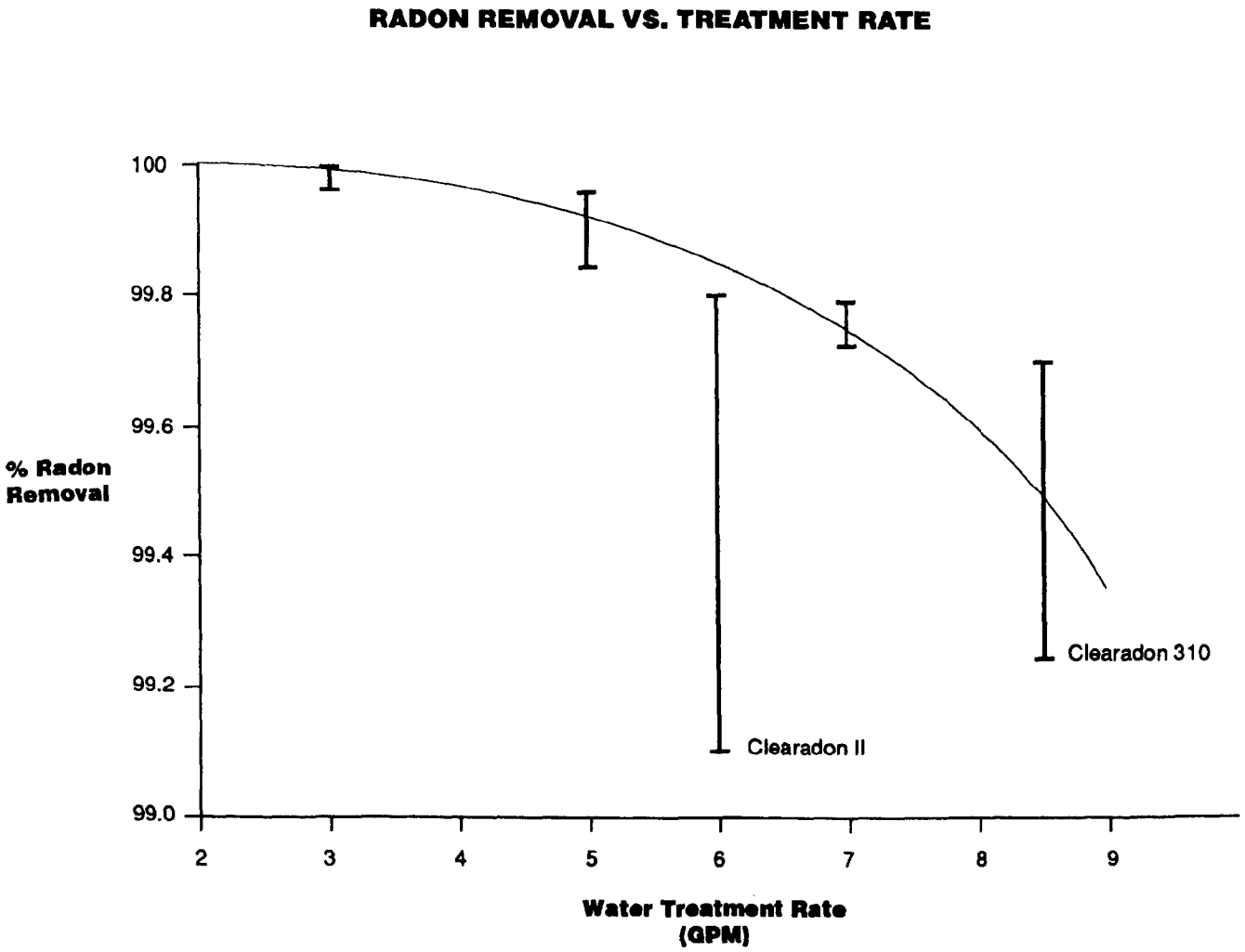

Figure 5

Figure 5 shows the relationship between treatment efficiency and water flow rate through the equipment. As one would expect, the percentage reduction of radon decreases with increasing flow rate when all other factors are held constant.

CUSTOMER CONCERNS

Other than the mechanical problems discussed earlier, the principal concern of our customers has been with the acceptability of any radon treatment device to future buyers of the home. Since radon and radon treatment equipment are unfamiliar topics to most people it is understandable that this would be a common concern. We have found that in a small percentage of cases (less than 10%) this does discourage a potential customer from purchasing the equipment. Of course it is quite likely that many people with this concern never contact us in the first place. I believe that our experience under estimates the actual proportion of people that have this concern.

TRAINING

Dealer training is one of the most important aspects of distributing treatment equipment such as this. To a person familiar with water treatment and water treatment equipment, installation and maintenance of a Clearadon unit is very straight forward and simple. To those unfamiliar with these areas there is a great deal to learn. We have therefore set up training courses for all of our dealers. Prior to installing any units we now require that the dealer's service people attend a one day training course at our factory. We are, of course, on call to our dealers to answer any questions about unusual situations as they come up. The equipment is normally sold only to our authorized and trained dealers. The equipment is not designed to be installed or serviced by the final consumer.

CURRENT STATUS OF THE RADON IN WATER INDUSTRY

In comparison to the size of the potential market for radon in water treatment equipment, the actual market is virtually nonexistant. There are approximately 18 million drilled private wells in the US. At a radon in water standard of 200 pCi/L approximately one half of these wells would require treatment. That is 9 million wells. In the past year between 150 and 300 aeration systems were installed. This low number of installations is due to the lack of a Maximum Contaminant Level standard for radon.

Representatives of the EPA have repeatedly stated that radon in water presents a **larger risk of death that all other water contaminants combined.** The risk posed is greater than the risks posed by lead, arsenic, hazardous waste site contamination and gasoline contamination of wells combined. Without a radon standard the vast majority of the population will not know about radon contamination of water and will not treat their water supplies. By EPA's own estimates, between 1000 and 1800 people die each year due to exposure to radon originating in their water supply.

The current market for radon water treatment equipment is currently limited to the exceptionally health conscious, wealthy population and to some real estate transactions. The market is evenly split between these two market segments.

The standard will be established for public drinking water supplies only since the EPA and other federal agencies do not have clear mandates to regulate environmental matters in private homes. However, it is obvious that if a standard is set for public water supplies it is prudent for individual homeowners to follow the same standards for their own protection. It is also possible that state Public Health Departments may, once a federal standard is set, establish their own water supply standards similar to those for bacterial contamination of private wells.

CURRENT STATUS OF THE RADON IN WATER STANDARD

Under the 1986 amendments to the Safe Drinking Water ACT (SDWA), the EPA is required to publish Maximum Contaminant Level Goals (MCLGs) and promulgate National Primary Drinking Water Regulations (NPDWRs) with respect to 83 specific drinking water contaminants according to a specified timetable. This original list of 83 contaminants included radon. Standards for all 83 contaminants were specified to be issued by June 19, 1989. When MCLGs are published, EPA must also propose Maximum Contaminant Levels (MCLs) that must be set as close to the MCLGs as is "feasible". The term "feasible" means with the use of the best technology , treatment techniques and other means, which the administrator of EPA, after examination for efficiency under field conditions and not solely under laboratory conditions, are available (taking costs into consideration).

On September 30, 1986, EPA published an advanced notice of proposed rulemaking. In that notice it was stated that the health hazard posed by radon contamination of water supplies is probably larger than that posed by all other water contaminants.

A Fact Sheet issued by the Criteria and Standards Division of the EPA Office of Drinking Water in September of 1988 stated that MCLGs and MCLs for radon would be proposed in September of 1989 and projected final regulations in November of 1990. An EPA Office of Drinking Water publication issued in May of 1989 stated that EPA is considering setting MCLs for radon in the range of 200 to 2,000 pCi/L of water. This publication continued to delay the ultimate regulation of radon stating that the "publication of the Notice of Proposed Rulemaking in the Federal Register is presently scheduled for June 1990." We have been informed that the Office of Drinking Water has all of the technical information they need in order to propose a radon standard. The only issues holding up the standard are policy and legal questions. As of this writing, the standard has been delayed again, to September of 1990.

EPA estimates that between 1000 and 1800 people die of lung cancer in the US each year from radon contamination of well water. Between June of 1989 and September of 1990, between 1200 and 2200, people have died because of the agency's inaction. Unnecessary deaths will continue until the EPA meets its Congressionally mandated responsibilities and proposes a radon standard. Only then will people be made aware of the danger and begin to take corrective measures.

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National Residential Radon Survey

by: Frank Marcinowski
U.S. Environmental Protection Agency
Office of Radiation Programs
Washington, DC 20460

Robert M. Lucas
Research Triangle Institute
Research Triangle Park, NC 27709

Abstract

The Superfund Amendments and Reauthorization Act (SARA) requires the EPA Administrator to conduct a national assessment of radon levels where people normally live and work, including educational institutions.

The National Residential Radon Survey (NRRS) is the first comprehensive effort to estimate the frequency distribution of average annual radon concentrations nationwide. Also, the survey will provide data to correlate radon concentrations with construction characteristics.

A stratified three stage area probability sample was used to randomly select approximately 12,000 homes. A questionnaire will provide information on living patterns, house construction, and heating, ventilation, and air conditioning (HVAC) characteristics. Two to four alpha-track detectors were placed in each home. It is expected that approximately 5,000 residents will return detectors with readable radon concentrations.

With this data, EPA will be able to accurately estimate the magnitude of the radon problem and evaluate the need for subsequent radon programs.

This paper has been reviewed in accordance with the U.S. Environmental Protection Agency's peer review and administrative review policies and approved for presentation and publication.

Introduction

The National Residential Radon Survey (NRRS) is mandated under Section 118(k) of the Superfund Amendments and Reauthorization Act of 1986 (SARA, P.L. 99-499) which requires the Administrator of the Environmental Protection Agency (EPA) to conduct a national assessment of radon "found in structures where people normally live and work, including educational institutions". The NRRS is addressing the component of the national distribution of radon concentrations in occupied housing units.

The results of the NRRS will be used to accurately estimate the magnitude of the national radon problem, assess the effectiveness of various radon program strategies (i.e., government/industry capability to address the radon problem, magnitude of State and Regional assistance, health risks) and evaluate the need for subsequent radon programs.

Intended Uses of the Data

The primary uses of the survey data will be to estimate the distribution of annual average radon concentrations in occupied housing units and the correlations among specific housing structures and radon concentrations.

In addition to estimating the national distribution of radon concentrations, EPA also plans to study subsets of the surveyed homes to learn how radon levels vary among smaller segments of the population. A representative sample of homes selected from each of the 10 EPA Regions will allow estimation of the distribution of radon concentrations in each region and also highlight differences in radon levels across the country.

Comparison of radon concentrations will also be made among other subgroups of the study population. The radon distribution of single family homes will be compared with multi-unit homes, rental units will be compared to owner occupied units, and radon levels will be compared among floors within the same structure to demonstrate how radon concentrations in basements relate to other levels of a home.

A secondary benefit of the NRRS includes collecting data related to health risks. The survey will give information on the concentrations of radon to which people are actually exposed to in their homes, as well as collecting data pertaining to the living habits of the participants. The survey will investigate homes with smokers, homes without smokers, homes with children, and various other aspects. Although residential

exposure is only a fraction (60-75%) of a person's total exposure, this information should help EPA estimate the fraction of lung cancers that can be averted through reduction of radon concentrations in homes.

Sample Design

A well designed scientific survey must have a prespecified well-defined inference or target population and specified informational requirements on which to base the sample design. The target population of the NRRS is:

a) All Census-defined housing units that are continuously occupied for 12 months that are not on military installations and

b) the permanent residence of these housing units.

The specified informational requirements are given in terms of the required relative precision of estimates of the upper tails of the distribution of annual-average radon concentrations in the U.S. The objectives used to develop the sample design are given below.

The first priority of the NRRS is to furnish a scientifically competent estimate of the national frequency distribution of annual average radon concentrations in occupied residences. Specifically, the estimate of the percent of homes with radon concentrations greater than 10 pCi/l should have a relative standard error no greater than 0.5, if the estimate of the percent of homes with radon concentrations greater than 10 pCi/l is approximately 0.5%.

The second priority of the NRRS is to furnish estimates of the frequency distribution of annual average radon concentrations in occupied residences for each of the 10 EPA Regions. The survey should also provide information from which correlations can be made between radon concentrations, house construction, and HVAC characteristics. Specifically, the estimate of the percent of homes with radon concentration greater than 4 pCi/l should have a relative standard error of no more than 0.5 for an EPA Region, if the estimate of the percent of homes with radon concentrations greater than 4 pCi/l is approximately 7%.

The NRRS is divided into two phases. Phase I of the survey involved identifying the households to be sampled, interviewing the residents of each household, and placing the detectors. Phase II includes several periods of panel maintenance, collection of the detectors, analysis of the data, and production of a final report.

Phase I

The survey population contains only residential households whether detached or multi-unit, single or multi-family, owner occupied or rental. A three stage area probability sampling strategy was used to select the housing units for the survey. The model assumed that the percentage of homes above 4 and 10 pCi/l varied among each EPA region and also across major geographic regions within a single EPA region. Each of the EPA defined regions was divided into substrata depending on the predicted percent of homes with a specific radon potential (Table 1). Differences between the assumed distribution in Table1 and the true distribution will not induce bias in the survey estimates but only reduce the precision of the estimates. A probability sample of primary sampling units (PSUs) was selected from each of the 22 defined strata with a minimum of 2 PSUs per stratum.

Table 1. Assumed Radon Potential Criteria Used to allocate the sample to Substrata.

Predicted Percent of Homes > 4 pCi/L	Predicted Percent of Homes > 10 pCi/L	Substratum Ranking
13	1.0	High
7	0.2	Medium
1	0.1	Low

The secondary sampling units (SSUs) within each PSU consisted of Census defined blocks or enumerations districts. Each of the SSUs contained at least 25 but no more than 150 housing units (HUs). Eight SSUs, with a probability proportional to the number of occupied HUs in the SSU, were selected within each PSU. Each SSU was visited by experienced field representatives who counted and listed the HUs within each of the 1,000 SSUs. Of these 1,000 SSUs, seven were identified as being in military installations and these contained no eligible HUs.

A random systematic sample of a specified number of listed HUs was selected from each SSU. A total of 11,441 HUs was selected to be visited by the field interviewers to determine participation in the survey, administer the NRRS questionnaire, and place the radon detectors. A total of 2,515 HU's were not eligible for the survey because they were vacant or not year-round occupied HUs (To meet the eligibility requirements of the NRRS, the residents had to have no firm plans to move within 12 months or must live within the residence for 9 months out of the year). Of the 8.926 eligible HUs, 80 percent (7,134) of the occupants fully cooperated by completing the questionnaire and having detectors placed in their homes. The targeted number of HUs expected at completion of the one year detector deployment period is approximately 5,000.

One of the primary data collection instruments of the NRRS is the household questionnaire which was administered by trained field interviewers. This questionnaire provided detailed information on the construction type and structure of the sampling residence, the amount of time spent on each level of the house, the type of HVAC system, smoking history of residents, occupancy habits, and detector placement information.

Following administration of the survey questionnaire, year long alpha-track detectors (ATDs) were placed in participating residences. Detectors were placed at each level of the home used as living areas up to three levels (excluding basement or crawl spaces). Single level homes received one detector in the living area and one in the bedroom area. Multilevel homes received a maximum of three detectors. Additionally, the basement received a detector if one wall or more was completely below ground. Any crawl space received a detector if it could be entered from inside the home, or could only be entered from outside the home and an adult could stand in it. Each residence received 2 to 4 detectors, with an average of 2.5 detectors per residence being placed for the survey.

Phase II

One of the principle tasks of Phase II is to conduct panel maintenance of the participating households. This will be accomplished through postcards and follow-up telephone calls for three periods during the yearlong detector deployment period. The purposes of these contacts are to verify the participation of the residents, confirm that they will continue to inhabit the same residence, convey a positive sense of participating in a valuable research venture, and determine if any problems exist with the detection devices.

The first round of panel maintenance has been completed with 0.7% of the residents no longer wanting to participate, 3.7% of the residents were moving and no longer were able to participate, 2.3% of the participants reported lost or damaged detectors, and 3.9% were not contacted after repeated attempts. The damaged, lost, and exposed detectors were replaced. Of the remaining eligible HUs, 95 percent (6,548) are still participating.

The detectors will be retrieved through the mail during the summer of 1990. The ATDs will be analyzed and the results will be sent to the participants in the survey. Various statistical analysis will be performed on the data to meet our primary and secondary objectives.

NRRS Schedule

Developed Survey Design	Summer 1988
Pretest Data Collection	February 1989
Administer Questionnaires and Place Detectors	June-August 1989
Panel Maintenance	September-October 1989 December 1989 - January 1990 March-April 1990
Retrieve Detectors	June-September 1990
Complete analysis of data and Announce Results	Spring 1991

Possible Data Analysis Strategies

The data from the NRRS will be primarily analyzed using descriptive statistics. The radon concentrations will be summarized by arithmetic and geometric means and standard deviations. These statistics will be determined for the entire nation, single unit homes (attached and detached), and multi-unit homes. The distribution of these concentrations will be examined to determine if it is lognormal.

The data will also be analyzed on a regional basis. This analysis will include all the residences sampled or could be limited to single unit homes. The sample size of the multi-unit homes is sufficient to provide a national estimation but is too small to provide regional estimations. The analyses will probably be based on the living level radon concentrations but could also be presented for basements and second floors.

The analyses will also describe the radon characteristics in single and multi-unit structures. For single unit structures, the radon characteristics of different types of crawl spaces (covered, uncovered), basements (storage, workroom, living, sleeping), first floors (full basement, partial basement, crawl space, slab on grade), and second floors will be examined. The multi-unit structures will be analyzed according to the number of units at ground level, below ground, and above ground. An important aspect of this investigation will be studying the relationship of radon concentrations to substructure characteristics.

The effect of energy conservation methods, designed to reduce air infiltration and heat loss, on radon concentrations will be examined. The housing population can be broken down according to slab on grade, basement, and partial basement/ crawl space, and the type of energy conservation method utilized.

Other analyses can be made between the radon concentrations and the type of HVAC system, age of the structure, degree of ground contact, and portion of the year closed to outside air. The data could also be analyzed according to the 11 regional geologic provinces, as outlined by the U.S. Geologic Survey, and comparisons made between the geologic characteristics and the radon concentrations within a given region.

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