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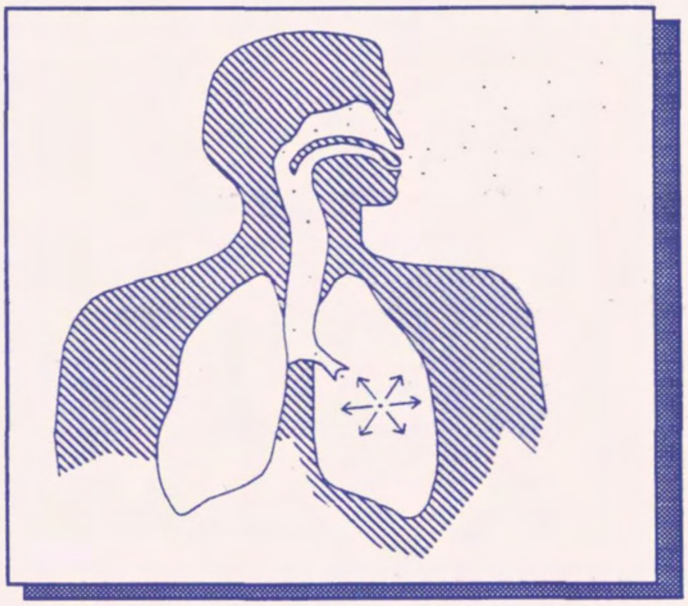

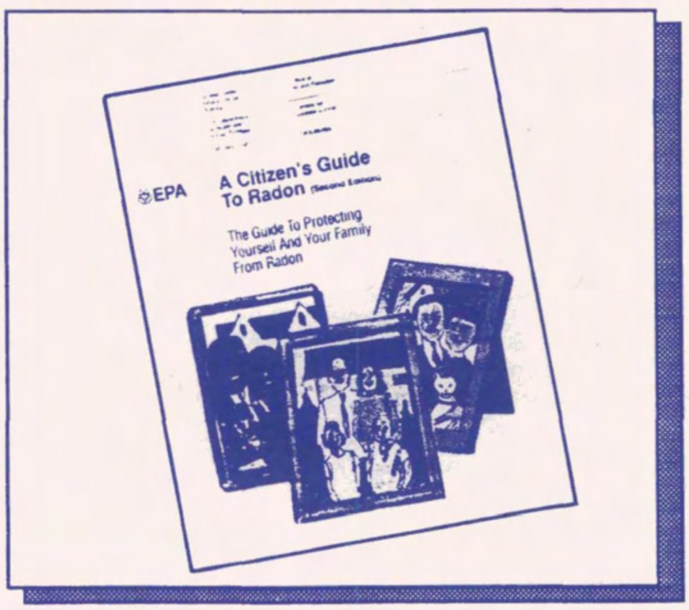

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TECHNICAL SUPPORT DOCUMENT FOR THE 1992 CITIZEN'S GUIDE TO RADON

Radon Division
Office of Radiation Programs
U.S. Environmental Protection Agency

May 20, 1992

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The efforts of several additional key contributors within ORP were instrumental in the development of this document, including Mike Walker (Project Director of the 1992 *Citizen's Guide*), Jerry Puskin, Chris Nelson, Anita Schmidt, Marion Ceraso, Dennis Wagner, Mark Dickson, Lisa Ratcliff, Stacy Greendlinger, Jed Harrison, Kirk Maconaughey, Dave Rowson, and Steve Page.

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Finally, ICF Incorporated assisted ORP in preparing this document. The ICF project team was directed by Sam Napolitano and Steve Wyngarden, and included Kerry Boshes, Freddi-Jo Eisenberg, Sarah Stafford, and John Trever.

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CHAPTER 1

INTRODUCTION AND BACKGROUND

This document presents the wide range of technical analyses, radon risk communication research, legislative directives, and other information that the U.S. Environmental Protection Agency (EPA) used to shape the policies that are set forth in the 1992 *A Citizen's Guide to Radon*. The document summarizes extensive technical analyses of the data that have been gathered over the past six years.

Background on the 1986 *Citizen's Guide*

In late 1984, the discovery of extremely high indoor radon in the Reading Prong area of Pennsylvania, New Jersey, and New York thrust radon into the national spotlight as a major public health problem. The government quickly responded to this problem by creating the Radon Program in 1985 to help States and homeowners reduce the health risks of indoor radon. EPA's Radon Program is non-regulatory, designed to provide public information and technical assistance to enable citizens to make informed decisions on how they should protect themselves.

A Citizen's Guide to Radon has played a major role in the overall strategy of the Radon Program. It serves as both EPA's core policy statement and primary public information brochure on radon in homes, as well as the Federal government's principal guidance document on indoor radon. Published jointly with the U.S. Department of Health and Human Services (HHS) in 1986, the original *Citizen's Guide* provided basic background information on indoor radon and its associated health risks, explained how radon levels in homes should be measured, and advised homeowners on when they need to take action to reduce radon levels in their homes.

When EPA and HHS issued the 1986 *Citizen's Guide*, there was less information on indoor radon than is available now. The recommendations in 1986 were based primarily on limited experience with homes whose indoor air had been contaminated with radon from uranium mill tailings. Research in these homes showed that most could be mitigated consistently to an average indoor radon level of 4 picocuries per liter (pCi/L). Based on this technology limitation, EPA established 4 pCi/L as the action level at which people should fix their homes.

Since little information was available from many homes on the relationship between short-term and annual average radon concentrations, and because the scientific community agreed that long-term measurements provided a better estimate of a home's annual average radon level, it was important to stress the need for long-term measurements before decisions on the need for mitigation were made. The Agency also wanted to provide States and homeowners with the means to gauge worst-case radon levels existing in their home or area, so that, the need for further action could be determined without waiting 12 months for the results of the measurement. These two considerations led to the development of a two-step strategy of measuring radon: (1) a short-term screening measurement in the lowest livable level during the wintertime, intended to measure the highest concentration to which occupants could be exposed; and (2) if the results of the screening measurement were above 4 pCi/L, a confirmatory measurement in lived-in levels, designed to provide an estimate of the average radon levels to which the occupants were exposed. The recommended duration of the confirmatory measurement depended on the results of the screening measurement and ranged from less than one week to one year.

Factors Influencing Revisions to the *Guide*

Many factors influenced the development of the revised *Citizen's Guide*. These factors included:

- The Indoor Radon Abatement Act, passed by Congress in 1988, which requires EPA to issue a revised *Citizen's Guide*, and contains specific requirements for the new *Guide*;

- Advancements in the understanding of the health risks associated with exposure to residential radon;

- Improved understanding of the accuracy of measurements made under different conditions for purposes of determining the need for mitigation;

- Advancements in the understanding of mitigation technology;

- Analyses of the costs and benefits of alternative action levels and testing strategies; and

- Risk communication research findings.

These factors are briefly reviewed in the following sections and are covered in detail in the remaining chapters of this document.

Indoor Radon Abatement Act (IRAA)

This amendment to the Toxic Substances Control Act endorses the non-regulatory strategy of the Radon Program and provides EPA with funding and a legislative mandate for a variety of additional activities designed to further reduce the risks of indoor radon exposure. Section 301 of the Act establishes the national long-term goal that indoor air be as free of radon as ambient air outside of buildings. This goal reflects the Congressional direction contained in Report language that EPA be more protective of public health. Results from EPA's Ambient Radon Field Study, initiated to comply with this requirement and confirm reported outdoor radon levels, are summarized in Appendix A.

Congress provided specific guidance on the scope of the revised *Citizen's Guide* in section 303, requiring EPA to provide a "series of action levels with a description of the associated health risks" to ensure that the public understands that 4 pCi/L is not a safe level of exposure and that a significant health risk exists at levels below 4 pCi/L. The Act also requires EPA to address health risks to special populations, i.e. children and smokers. In addition, Congress specified that EPA present an analysis of the relationship between short- and long-term measurement results and their relative impacts on alternative action levels.

Updated Radon Risk Information

The Biological Effects of Ionizing Radiation (BEIR IV) Committee of the National Academy of Sciences issued a report (NAS 1988) on the effects of exposure to alpha radiation that includes extensive information on the results of studies of underground miners exposed to radon decay products. National radiation risk experts comprising the BEIR IV Committee prepared and reviewed a radon risk projection model (the BEIR IV model). EPA used the BEIR IV model to estimate radon risk, making slight adjustments to the model to account for background radon and to incorporate the most recent information on dose per unit exposure in mines and homes (NAS 1991). The Agency also (1) used new data on residential radon levels collected through EPA's National Residential Radon Survey (U.S. EPA/Office of Radiation Programs 1991a); (2) assembled an independent panel of experts on radiation risk (the Agency's Science Advisory Board), who reviewed the Agency's risk calculation methods and provided EPA with further suggestions for refinements; and (3) conducted a detailed analysis of uncertainty associated with the radon risk estimates. EPA's revised risk estimates fall within the ranges established by national and international scientific organizations, including the National Academy of Sciences, the International Commission on Radiological Protection, the National Council on Radiation Protection and Measurement, and the United Nations Scientific Committee on the Effects of Atomic Radiation. Chapter 2 discusses EPA's risk assessment methodology and risk estimates in detail. Appendix B provides basic background information on radon and its decay products.

Analysis of Measurement Options

Since publishing the 1986 *Citizen's Guide*, EPA has conducted extensive research to determine how well various radon testing options meet the basic purpose of testing -- to enable homeowners to make the right decision on whether to mitigate radon levels. The Agency drew from the wealth of radon testing data and experience gained since 1986 to develop a number of testing options, each defined by a given action and "trigger level," a certain testing location, appropriate ventilation conditions, and different measurement durations. Using a statistical model, data on the magnitude of measurement errors caused by various sources, and the distribution of annual average radon levels developed from the National Residential Radon Survey, EPA then evaluated each testing option in terms of how frequently it would result in homeowners making a correct mitigation decision (i.e., how well a testing procedure provides a reasonably accurate basis for determining whether a home is above the action level). EPA considered this information on the effectiveness of testing options, together with real-world information on public acceptability, in order to select a revised testing protocol for the 1992 *Citizen's Guide*. Chapter 3 provides the details of this analysis and Appendix C provides background information on commonly used radon measurement devices.

Advancements in the Understanding of Mitigation Technology

There has been progress in the understanding of radon measurement and mitigation technology since 1986, with a wide variety of methods now available to prevent radon entry into homes and to reduce indoor radon concentrations after entry. A radon mitigation industry also has emerged to provide homeowners throughout the country with the ability to reduce their radon levels. Research by EPA's Office of Research and Development and a recently completed survey of radon mitigation contractors (U.S. EPA/Radon Division 1990a) have shown that radon mitigation in the vast majority of cases should leave homes with an annual average level of less than 4 pCi/L. It also will be very common for homes to have post-mitigation levels of 2 pCi/L or less. Chapter 4 describes current mitigation technology and its overall effectiveness and cost.

Analysis of Costs and Benefits

Additional data collected since the *Citizen's Guide* was first published have enabled the Agency to conduct a detailed analysis of the costs and risk reductions (benefits) for the public if it followed the *Guide's* testing and mitigation advice. Based on this analysis, EPA has estimated (1) the costs of the different testing and mitigation options considered for inclusion in the revised *Guide* (i.e., the total annualized costs of testing and mitigation); (2) the risk reductions provided by the different options (i.e., the annual lives saved by testing and mitigation); and (3) the cost-effectiveness of alternate approaches (i.e., the cost per lung cancer death averted). EPA also has evaluated the cost-effectiveness results by relating them to the public's willingness to pay to save a statistical life and by comparing the estimated cost per life saved for radon reduction to the cost per life saved for other health-related programs. The methods and results of this analysis are described in Chapter 5 and in Appendices D, E, F, G, and H.

Risk Communication Research

Substantial information is now available on risk communication and, in particular, how the public responds to radon risk messages. Researchers and public health officials report that homeowners do not overreact or panic after receiving information on radon. Instead, the public remains largely apathetic about radon, as evidenced by the small proportion of homes that have been measured and fixed. Radon risk communication research has provided useful insight into why this apathy exists and identified six key ways for public information materials to help overcome it. One of the most important findings has been that the guidelines on testing and mitigation presented in the 1986 *Guide* need to be streamlined in order to minimize barriers to public action. Homeowners often drop out of the two-step testing process, especially if it means having to wait for an entire year to get the results of the recommended long-term follow-up measurement. A 1990 telephone survey (Johnson 1990), for example, found that only 9 percent of the participants were willing to conduct a one-year radon test. In fact, it appears that those homeowners who do mitigate are acting on the basis of short-term measurements in the lowest livable area of their home. Current research on radon risk communication and a summary of how that research has been incorporated into the revised *Guide* is provided in Chapter 6.

CHAPTER 2

ESTIMATION OF RISKS

Section I: Overview

Radon has been classified as a known human carcinogen based on extensive data from epidemiologic studies of underground miners.[1] The carcinogenicity of radon has been well established by the scientific community, including the World Health Organization's International Agency for Research on Cancer (IARC 1988), the Biological Effects of Ionizing Radiation (BEIR IV) Committee of the National Academy of Sciences (NAS 1988), the International Commission on Radiological Protection (ICRP 1987), and the National Council on Radiation Protection and Measurement (NCRP 1984). In addition, the Centers for Disease Control, the American Lung Association, the American Medical Association, and the American Public Health Association have recognized radon as a significant public health problem. Radon has been characterized as "a potentially important cause of lung cancer in the general population, which is exposed through contamination of indoor air by radon from soil, water, and building materials" (Samet 1989).

> THE AIMS OF THIS CHAPTER
>
> (1) To document EPA's approach for estimating the lung cancer risk attributable to radon exposure.
>
> (2) To provide information critical to the interpretation of the risk estimates, including information on uncertainty and the effect of smoking.

To assess the risk of lung cancer associated with residential exposure to radon, EPA currently uses the risk projection model developed by the BEIR IV Committee (NAS 1988) with minor modifications by EPA. EPA adjusted the BEIR IV relative risk model to incorporate the most recent information on dose per unit of exposure in mines and homes from the report: *Comparative Dosimetry of Radon in Mines and Homes* (NAS 1991), and to account for background radon. EPA's current estimate of risk incorporates the estimate of the average national residential radon level from the recently completed EPA National Residential Radon Survey (U.S. EPA/Office of Radiation Programs 1991a). Additionally, the Agency has included in this radon risk assessment a detailed analysis of the uncertainty associated with the lung cancer risk estimates, based on the approach employed in the National Institutes of Health report on the development of radioepidemiological tables (NIH 1985).

Using the modified BEIR IV model, EPA estimates that the number of lung cancer deaths per year in the U.S. due to residential radon exposures is approximately 14,000, with an uncertainty range of 7,000 to 30,000. This estimate is based on the Census Bureau's estimate of 250 million people for the residential U.S. population as of October 1, 1990 (Bureau of the Census 1992a).

[1] The term radon when used in this chapter refers to radon-222 and its decay products. Of the short-lived radon decay products (polonium-218, lead-214, bismuth-214, and polonium-214), the polonium isotopes contribute most of the radiologically significant dose to the lung. The risk from inhaled radon-222 is small compared to the risk from inhaled radon-222 decay products.

EXHIBIT 2-1
ESTIMATES OF LUNG CANCER RISK FROM EPIDEMIOLOGICAL STUDIES OF UNDERGROUND MINERS EXPOSED TO RADON

Study Population	Average Exposure (WLM)	Relative Risk Coefficient (%/WLM)[a]	Reference
Czech Uranium Miners	313 226	1.92 1.5	Thomas et al. 1985 Sevc et al. 1988
Ontario Uranium Miners	40-90	0.5-1.3 1.4[b]	Muller 1984 NAS 1988
New Mexico Uranium Miners	111.4	1.8	Samet et al. 1991
Swedish Iron Miners (Malmberget)	81.4	3.6 1.4[b]	Radford & St. Clair Renard 1984 NAS 1988
Colorado Plateau Uranium Miners	834	.45 0.6[b]	Thomas et al. 1985 NAS 1988
Eldorado (Beaverlodge) Uranium Miners	20.2	3.28 2.6[b]	Howe et al. 1986 NAS 1988
Newfoundland Fluorospar Miners	382.2	0.9	Morrison et al. 1988

[a]The relative risk coefficient is the fractional increase above the baseline lung cancer incidence or mortality rate per WLM.

[b]Estimate based on reanalysis of the data by the NAS with the cooperation of the principal investigators.

- When cumulative exposures were equal, low exposures over longer periods produced greater lung cancer risk than high exposures over short periods.

- Increased lung cancer risk with radon exposure has been observed even after controlling for, or in the absence of, other mine exposures such as asbestos, silica, diesel fumes, arsenic, chromium, nickel and ore dust.

- Increased lung cancer risk has been observed in miners at relatively low cumulative exposures (e.g., 40-70 working level months[2] (WLM) in the study of Ontario uranium miners).

- Non-smoking miners exposed to radon have been observed to have an increased risk of lung cancer.

In their review of five of the major studies of underground miners, Samet and Hornung (1990) noted that although there were significant differences in the studies, all demonstrated an exposure-response relationship between exposure to radon and excess lung cancer, and excess relative risk per WLM estimates that were "remarkably homogeneous." The following sections review findings from eight major studies, including those reviewed by Samet and Hornung. This is not meant to be an exhaustive review, but rather represents an overview of the findings from a number of the prominent studies of miners. The studies reviewed here were conducted in Czechoslovakia, Ontario (Canada), New Mexico (U.S.), Sweden, the Colorado Plateau (U.S.), Australia, Northern Saskatchewan (Canada), and Newfoundland (Canada). The miners in these studies continue to be followed over time, and updated analyses are conducted as new data become available.

The Czechoslovakian study included four large cohorts, with a total of 9,403 uranium miners (Samet and Hornung 1990). Average cumulative exposures for the cohorts ranged from 3.2-303 WLM, and average length of follow-up varied from 6-30 years. Excess lung cancer mortality was found largely in the cohorts with the highest cumulative exposures and the longest follow-up periods. However, the study also detected a significant increase in the risk of lung cancer from levels of cumulative exposure as low as 50-99 WLM. The attributable risk of lung cancer was observed to increase with age at initial exposure, and the overall attributable risk was estimated to be 20 per WLM per 10^6 person-years. When cumulative exposures were equal, low exposures over long periods were found to produce higher risk than high exposures over short periods. The combined effect of cigarette smoking and exposure to radon progeny was observed to be approximately additive. Strengths of this study include the extensive exposure data and long follow-up periods (25-30 years for two of the four cohorts). The study is limited by incomplete smoking information for the cohorts that had the highest exposure and the longest follow-up, and by the use of analytical methods that did not readily allow for comparison with other studies.

The Ontario, Canada study included 15,984 uranium miners, a large number of whom were exposed at relatively low levels (Samet and Hornung 1990). The mean cumulative exposures of the miners in this study were 40 WLM and 90 WLM, depending upon the method used to estimate exposure, and the mean period of follow-up was 15 years. At exposure levels of 40-70 WLM, lung cancer mortality was found to be significantly increased. The attributable risk of lung cancer was 7 or 3 per WLM per 10^6 person-years, again depending upon the exposure estimate. The excess relative risk was calculated to be 1.3% per WLM. The study also showed a decrease in risk with time since exposure. Strengths of this study are the large number of miners exposed to radon progeny at relatively low levels, and the consideration of exposures received in other types of hard-rock mining. The study is currently limited by the fact that the members of the cohort were still too young at the end of the follow-up period (median age was 49 years) for observation of the complete temporal

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[2]A working level is defined as any combination of short-lived radon decay products in 1 liter of air that will result in the ultimate emission of 1.3×10^5 MeV of potential alpha energy. A working level month is defined as the exposure to 1 working level for 170 hours (1 working month).

expression of excess lung cancer risk from exposure to radon progeny. There was also a lack of smoking information for study subjects.

The New Mexico study included 3,469 uranium miners who had worked at least one year underground in New Mexico uranium mines prior to December 31, 1976 (Samet et al. 1991b). The follow-up period extended through 1985, and the mean cumulative exposure level for the cohort was 111.4 WLM. The risk of lung cancer in this cohort was increased for exposure categories above 100 WLM; the excess relative risk increased by 1.8% per WLM. The study data were consistent with a multiplicative interaction between radon and smoking. Additionally, lung cancer risk was observed to increase more steeply for those less than 55 years of age. The authors noted that the study may suffer from random misclassification of radon exposure, and possible systematic bias in exposure estimates. However, the authors cite the extensive database on which exposure estimates were based as one of the strengths of the study.

The Swedish study included 1,415 iron miners, with an average cumulative exposure of 81 WLM (Samet and Hornung 1990). This study had the longest follow-up period of the eight studies; the average time in the study was 44 years. Of the 1,415 miners, 1,294 were observed between 1951 and 1976, during which time 50 lung cancer deaths occurred. Based on Swedish national mortality rates, 14.6 lung cancer deaths were expected. The study found significant excess risk for cumulative exposures above 80 WLM, and attributable risk of lung cancer was estimated to be 19 per WLM per 10^6 person-years. Through an analysis that required many assumptions, the combined effect of smoking and exposure to radon progeny was estimated to be additive. Silica, diesel exhaust, arsenic, chromium, and nickel were discounted as potential confounding exposures because they were present in small concentrations in the mines. The strengths of this study are the length of the follow-up periods and the thorough confirmation of cause of death. The study is limited by uncertainty in the accuracy of exposure estimates, by the lack of smoking information for all subjects, and by the use of different methods of exposure assignment for lung cancer cases and non-cases.

The U.S. Public Health Service Colorado Plateau study included 4,127 uranium miners working in Colorado, New Mexico, Arizona, and Utah (Samet and Hornung 1990). Of these, 3,346 were included in the risk assessment conducted by the National Institute of Occupational Safety and Health. The average cumulative exposure level of this cohort was 821 WLM. The average length of follow-up was 22 years. In the range of exposure below 600 WLM, excess relative risk of lung cancer was found to be 1.2% per WLM, and attributable risk of lung cancer was 6.3 per WLM per 10^6 person-years. Relative risk was observed to decrease with time since cessation of exposure, and miners first exposed at older ages were at increased risk of lung cancer compared to those exposed at younger ages. As in the Czech study, low exposures over long periods of time were found to be more hazardous than high exposures for short periods of time, when cumulative exposures were equal. The relationship between smoking and exposure to radon progeny was estimated to be slightly submultiplicative. Strengths of this study are the large number of miners in the study, the close and lengthy follow-up, and the availability of smoking information for most study subjects. The high levels of exposure that characterize this study may limit its applicability to risk estimation at lower levels of exposure. Consequently, in their analysis of these data for the development of a radon risk model, the BEIR IV Committee excluded these very high exposure levels (above 2,000 WLM). In addition, although the most recent complete data analysis was based on smoking information collected in 1969, smoking histories have recently been updated and are being incorporated into a new analysis of the radon-lung cancer relationship in this miner cohort (Roscoe 1991).

The Colorado Plateau cohort was also the data source for a study of lung cancer risk in non-smokers exposed to radon (Roscoe et al. 1989). For this study, a sub-cohort of 516 male uranium miners who never smoked cigarettes, pipes, or cigars was selected from the original cohort of 4,127 miners, and was followed from 1950-1984. When the non-smoking miners were compared to non-smoking U.S. veterans, the miners were 12.7 times more likely to have died of lung cancer than were the veterans. When compared to male non-smokers in the general U.S. population, the miners were 9.3 times more likely to have died of lung cancer. No lung cancer deaths were found among non-smoking miners with less than 465 WLM of cumulative radon exposure. However, the limited number of non-smoking miners in the study who were exposed at lower levels severely restricted the statistical power of the study to detect risks at these lower exposures relevant to residential levels. The authors of this study concluded that: "exposure to radon daughters in the absence of cigarette smoking is a potent carcinogen that should be strictly controlled" (Roscoe et al. 1989).

The Australian study included 2,574 uranium miners (1,433 underground workers, 1,141 surface workers) at the Radium Hill mine which operated in eastern South Australia from 1952-1961 (workers were followed through 1987). Compared to other mines of that period worldwide, cumulative exposures of underground workers were low, with a mean of 7.0 WLM and a median of 3.0 WLM. When compared with the surface workers, lung cancer risk was significantly increased in the underground workers with exposures above 40 WLM. The authors concluded that it was unlikely that the increase in risk was due to differences in smoking habits or other confounders (such as asbestos, dust levels, diesel fumes). Strengths of the study include low exposure levels, a long follow-up period (more than 30 years for most workers) and relatively detailed historical information on radon gas concentrations at the mine. However, the authors noted that assumptions and approximations were necessary to estimate personal radon-daughter exposures. Additionally, according to the authors, the study was limited by the small size of the workforce and their inability to trace over one-third of the workers beyond the end of employment at Radium Hill.

The Eldorado (Beaverlodge) study included 8,487 workers employed at the Eldorado Resources Limited Beaverlodge Uranium Mine in Northern Saskatchewan (Howe et al. 1986). Follow-up was carried out from 1950-1980. The average cumulative exposure, weighted by person-years at risk was 20.2 WLM (NAS 1988). A highly significant linear relationship between dose and increased lung cancer risk was observed, with an estimated relative risk coefficient of 3.28% per WLM. This study is notable for the low cumulative exposure levels and for the low silica exposures and the absence of any diesel machinery in the mines. The authors note that the study was limited by the lack of any smoking information on the miners, and by incomplete information on the cohort's work experience at mines other than Beaverlodge where exposure to radon daughters may have occurred.

Subsequent to this study, a case-control study was undertaken within this cohort to examine the potential contribution of smoking and non-Beaverlodge work experience to the relative risk coefficients from the original Beaverlodge study (L'Abbé et al. 1991). The study included 46 lung cancer cases and 95 controls that were enrolled in the Beaverlodge uranium mine cohort study between 1950 and 1980. The authors concluded that cigarette smoking and other mining experience were unlikely to have made an important contribution to the relative risk coefficients reported in the original study. The study did not definitively distinguish between a multiplicative or an additive relationship between radon exposure and cigarette smoking, although the authors suggest that their data offer limited support for the multiplicative relationship.

The Newfoundland study included 1,772 Newfoundland underground fluorspar miners with an average cumulative exposure of 382.8 WLM, followed from 1950-1984 (Morrison et al. 1988). The study reported significantly elevated numbers of cancers of the lung, salivary gland and buccal cavity and pharynx. A highly significant association was observed between radon exposure and the risk of death from lung cancer. The relative risk coefficient was estimated at 0.9% per WLM, and those first exposed before age 20 had the highest relative risk coefficients. Smoking miners were observed to have relative and attributable risk coefficients comparable to those of non-smoking miners. The strengths of this study include the collection of information on cigarette smoking status for 48% of the cohort, and a long follow-up period (1950-1984). Study limitations include the lack of any radon measurements prior to 1960, and an inability to trace a large number of the workers (approximately 16% of the underground cohort as originally defined was excluded because of a lack of adequate personal identifying information).

Occupational data on exposure to radon and lung cancer risk continue to be made available through ongoing follow-up of existing miner cohorts, and through newly reported studies. The National Cancer Institute has acquired data from 11 cohort studies and in collaboration with the various principal investigators is undertaking a joint analysis. These studies represent all major cohorts of radon exposed miners currently available, and will include 2,300 lung cancer cases. The analysis is expected to be completed in 1992 (Lubin 1991).

Residential Studies

There is uncertainty associated with the projection of lung cancer risk from occupational radon exposures to the general population for residential exposures (see Section V for discussion of uncertainty). Residential studies have been undertaken in an attempt to directly define and examine the association between lung cancer incidence and exposure to radon progeny in the home.

There are a wide variety of residential radon epidemiologic studies that are already completed, ongoing, or being planned. Residential studies conducted to date can be divided into two basic categories, those that examine the association between radon and lung cancer based on group characteristics (e.g., ecologic studies), and those that examine the association based on individual characteristics (e.g., case-control studies).

Ecologic studies compare frequencies of exposure and disease between groups, rather than between individuals, or assess changes in disease frequency and exposure frequency over time for one or more groups. Selection of the group is often based on geographic parameters. Ecologic studies are relatively inexpensive and convenient, and are useful for generating hypotheses; however, they are not recommended for identifying the cause of a disease as they cannot: (1) determine the joint distribution of the disease (e.g., lung cancer) and the study factor (e.g., radon exposure) within each group, or (2) separate the effects of two or more variables (Kleinbaum et al. 1982). The use of ecologic (group) data to draw conclusions about effects in individuals (e.g., development of lung cancer) may yield causal inferences that suffer from an "ecological fallacy;" an association or the lack of an association between a study factor and disease that is observed at the level of the group may not hold at the individual level. The study groups being compared may differ in many factors other those being considered in the study, and any one of these may be the underlying cause of the difference in their disease states.

For ecologic studies of radon risk, the average radon exposure and average lung cancer incidence or mortality for each group can be identified. However, there is no way to relate the level of radon exposure for an individual to that individual's health status. The number of lung cancer patients with a history of elevated radon exposure or the number of individuals with elevated radon exposure who develop lung cancer is not determined. Because the unit of analysis is the group, this type of study is incapable of assessing individual smoking history or mobility, both of which are very important variables in radon risk assessment.

Completed ecologic studies have generally used characteristics of a geographical region, such as geology or radon in water, as surrogates for radon exposure and correlated them with lung cancer rates in the same area. Most of these studies showed associations between lung cancer incidence or mortality and exposure to radon. However, since individual exposures were not considered, results can only suggest that residential exposure to radon progeny increases lung cancer risk (Samet 1989).

The 1989 International Workshop on Residential Radon Epidemiology endorsed the use of case-control studies for residential radon research, and specifically recommended against the further use of ecologic studies because of their limitations for testing causal hypotheses (U.S. DOE/Office of Energy Research 1989).

Case-control studies involve the comparison, with respect to exposure, of individuals who have the disease of interest (cases) and those who do not (controls). Case-control studies are useful for testing causal hypotheses for specific diseases. They do not require extensive follow-up, and have a greater ability to control for potentially confounding variables (e.g., smoking, diet) than ecologic studies, since they collect information on study subjects individually.

Case-control studies have been used to more directly test the relationship between lung cancer incidence or mortality and residential radon exposure (Samet 1989). Early case-control studies, conducted in Sweden, the United States, and Canada, generally used housing characteristics as surrogates for radon exposure. A number of these studies found associations between the radon exposure surrogates and increased lung cancer incidence or mortality.

Four case-control studies of residential radon exposure and lung cancer risk have been completed recently. Unlike most of the earlier studies that used surrogates for exposure, these studies took measurements in the homes of study subjects to determine radon exposure. The results of these studies were mixed, with one study showing a statistically significant trend of increasing lung cancer risk with increasing radon concentration (Schoenberg et al. 1990), one study showing a borderline significant increase in lung cancer risk with estimated radon exposure (Pershagen in press 1992), and two studies showing no statistically significant association between radon exposure and lung cancer risk (Ruosteenoja 1991, Blot et al. 1990). The following is a brief review of findings from these four studies conducted in the U.S. (New Jersey), Sweden, Finland, and China.

A study of radon and lung cancer in New Jersey women with 480 lung cancer cases and 442 controls (Schoenberg et al. 1990) has reported a statistically significant trend of increasing lung cancer risk with increasing residential radon concentration. This trend was observed even after adjusting for smoking, age, and occupational history. Additionally, the observed increase in relative risk (3.4% per WLM), although associated with considerable uncertainty, is consistent with the increases reported in the studies of underground miners, supporting the extrapolation of data from the miner studies to the residential setting. However, the estimates of risk for the different categories of radon concentration, when taken individually, were not statistically significant. Furthermore, the authors of the study cite the small number of subjects with high radon exposures, the possibility of selection biases, and the incomplete cumulative exposure assessments as reasons for exercising caution in interpreting the study results.

A study of Swedish women with 210 lung cancer cases and 400 controls has found a borderline significant increase in lung cancer risk with estimated radon exposure in women whose average radon level exceeded 4 picocuries per liter (pCi/L). Positive trends were observed correlating the relative risk of lung cancer with the estimated cumulative residential radon exposure. Adjustment for the percentage of time actually spent in the home reduced the evidence for a dose-response relationship. The authors state that their findings suggest that radon exposure is of importance for the risk of developing lung cancer, particularly in the younger age groups. However, they cautioned that the small sample size limited the ability of the study to discriminate between various models in describing the association between estimated radon exposure and lung cancer risk (Pershagen et al. in press 1992).

A study of Finnish men with 291 lung cancer cases and 495 controls found no statistically significant relationship between indoor radon exposure and the risk of lung cancer (Ruosteenoja 1991). The exposure-response curve initially showed an increase but then deviated downward at the higher exposure levels (above 7.4 pCi/L). The authors postulated that this may have been due to random variation or systematic bias. The authors also noted that the final sample size was too small to detect the risk that was expected based on findings from the miner studies.

A study of Chinese women, which included 308 lung cancer cases and 356 controls, showed no statistically significant association between radon exposure and lung cancer risk, regardless of cigarette smoking status (Blot et al. 1990). A non-significant trend of increased lung cancer risk with increased radon exposure was observed among residents that were exposed to high levels of radon. The authors of the study advised cautious interpretation of these results, however, given the potentially important confounding effect of high indoor air pollution in the study area. Another issue, in light of the ethnic and cultural differences between China and the U.S., is the applicability of these results to the U.S. population.

A number of residential radon case-control studies that are presently in progress were presented at the International Workshop on Residential Radon Epidemiology in July of 1989 in Alexandria, VA (Samet et al. 1991a). The principal investigators of these studies as well as other scientists discussed a variety of issues at the workshop, including the issue of sample size. Workshop attendees concluded that the majority of the residential radon studies, when taken individually, have samples too small to characterize the exposure-response relationship between radon and lung cancer. As indicated in an analysis conducted by Lubin et al. (1990), the number of lung cancer cases needed to characterize the exposure-response relationship between indoor exposure to radon and lung cancer risk may be in the thousands. For this reason, the study investigators have decided to undertake a pooling of results from the international case-control residential radon studies as they are completed. By pooling the data and thus increasing sample size, the investigators hope to characterize the exposure-response relationship. The Second International Workshop on Residential Radon Epidemiology, held in Alexandria, VA in July of 1991, determined that an initial pooling effort would not be possible until 1994. Exhibit 2-2 summarizes 11 ongoing case-control studies of residential radon exposure and lung cancer risk.

EXHIBIT 2-2
ONGOING CASE-CONTROL STUDIES OF RESIDENTIAL RADON EXPOSURE AND LUNG CANCER RISK

Study Site	Principal Investigator	Funding Agency	Estimated Completion	# Lung Cancer Cases	Duration of Radon Measurement
U.S./CN	Jan Stolwijk	National Institute of Environmental Health Sciences	1993	1000	1 year
U.S./FL	Heather Stockwell	National Cancer Institute	1992	300	1 year
U.S./MO	Michael Alavanja	National Cancer Institute	1993	600	1 year
U.S./NJ	William Nicholson	National Institute of Environmental Health Sciences	1994	500	1 year/4 days
U.S./UT & So. ID	Joseph Lyon	National Institute of Environmental Health Sciences	1994	750	1 year/6 months
Belgium	Andre Poffyn	National Fund for Science Research	1994-96	100	6 months
Canada	E. G. Letourneau	Canadian Health and Welfare	1993	750	6 months
France	Margot Tirmarche	University of Brest/National Institute of Health and Medical Research	1994-96	600	3 months
Sweden	Goran Pershagen	National Institute of Radiation Protection and the Swedish Cancer Society	1992	1500	3 months
U.K.	Sarah Darby	Imperial Cancer Research Fund/National Radiological Protection Board	1994	600	6 months
West Germany	H. Erich Wichmann	Federal Ministry of Environment	1996	3200	1 year

Source: Second International Workshop on Residential Radon Epidemiology, Alexandria, VA, 1991.

A coordinated case-control study involving France, Germany, Luxembourg, the United Kingdom and Belgium is being sponsored by the Commission of European Communities. The coordinator is Dr. A. Poffyn at the State University of Ghent in Belgium.

Animal Studies

Further information on the deleterious health effects associated with exposure to radon has been provided by experimental studies of animals. According to the BEIR IV Committee, animal experiments conducted in both the United States and in France have confirmed the carcinogenicity of radon, and have provided insight into the nature of the exposure-response relationship, as well as the modifying effects of exposure rate (NAS 1988). The following is a summary of relevant findings to date from animal studies:

- Health effects observed in animals exposed to radon and radon decay products include lung carcinomas, pulmonary fibrosis, emphysema, and a shortening of life-span (U.S. DOE/Office of Energy Research 1988a).

- The incidence of respiratory tract tumors increased with an increase in cumulative exposure and with a decrease in rate of exposure (NAS 1988).

- Increased incidence of respiratory tract tumors was observed in rats at cumulative exposures as low as 20 WLM (NAS 1988).

- Exposure to ore dust or diesel fumes simultaneously with radon did not increase the incidence of lung tumors above that produced by radon progeny exposures alone (DOE/Office of Energy Research 1988a).

- Lifetime lung-tumor risk coefficients that have been observed in animals (between 1 and 5 x 10^{-4}/WLM) are similar to the lifetime lung-cancer risk coefficients observed in human studies (DOE/Office of Energy Research 1988a).

- In a study of rats exposed to radon progeny and uranium ore dust simultaneously, it was observed that the risk of lung cancer was elevated at exposure levels similar to those found in homes. The risk decreased in proportion to the decrease in radon-progeny exposure (Cross et al. 1991).

Section III: Exposure Assessment

EPA estimates annual U.S. lung cancer deaths due to radon by extrapolating risks observed in miners to the general public based on estimated cumulative radon exposures, expressed in terms of WLM. Exposure is often estimated indirectly, using assumptions or mathematical models. For radon, EPA has directly estimated exposures to the public based on the results of the National Residential Radon Survey (NRRS) (U.S. EPA/Office of Radiation Programs 1991a). The NRRS included approximately 6,000 randomly selected homes that were statistically representative of all residential structures across the fifty United States. Alpha track detectors were placed in each home for a period of one year, and a questionnaire was used to gather information on occupant living patterns, smoking habits, house construction, weatherization, and heating, ventilation, and air conditioning characteristics. The survey was implemented in 1989 and 1990 and completed in 1991.

An estimate of average exposure in the general population must include information on both the magnitude and duration of exposure. Based on results of EPA's NRRS, the current estimate of the average national residential radon concentration (i.e., magnitude of exposure) is 1.25 pCi/L (based on the average of all lived-in levels) (U.S. EPA/Office of Radiation Programs 1991a). Using this concentration, EPA has calculated the average annual U.S. cumulative exposure to radon to be 0.242 WLM. This estimate assumes that, on average, 75% of a person's day is spent in the home (i.e., duration of exposure), and the equilibrium factor is 0.5 (see section V.2.b for definition of the equilibrium factor).

The primary source of residential radon concentrations is soil gas which enters the house through cracks or openings in the foundation. However, radon can also enter the home through ground water sources, and be released into indoor air during household water use such as showering and washing clothes. It is estimated that this source accounts for approximately 5% of the total indoor air concentration in houses served by ground-water sources. On average every 10,000 pCi/L of radon in water contributes about 1 pCi/L of radon to indoor air. Generally, the contribution to lung cancer risk from radon that enters the home through water is small compared to the risk from radon that enters the home through soil gas. In addition, certain building materials and, to a lesser degree, natural gas may occasionally make a small contribution to indoor radon concentrations. This document focuses on the lung cancer risk associated with inhalation exposures to radon. Although the ingestion of drinking water contaminated with radon may add some minor risk of developing stomach or other internal organ cancer, this risk is small compared to the risks from breathing indoor air containing radon (U.S. EPA/Radon Division 1991).

There is some uncertainty associated with the estimate of the U.S. population's average annual cumulative radon exposure. The average residential radon level, the occupancy factor, and the equilibrium factor all contribute to the uncertainty in the estimation of average residential exposure. This uncertainty has been factored into EPA's overall estimate of risk as discussed in Section V.

Section IV: Characterization of Exposure and Response

The purpose of exposure-response assessment is to describe the relationship between exposure to the substance of interest and the occurrence of an effect or response (e.g., cancer). Subsequently, an estimate of the potential response at various levels of exposure can be made based on this relationship. Usually this means extrapolating from the effects observed at high levels of a substance administered to experimental animals, or at occupational exposures noted in epidemiologic studies, down to the generally lower levels of exposure to the general population.

Data on the relationship of exposure and response from human epidemiologic studies are clearly preferable to animal data when predicting risks to human populations. Uncertainties associated with applying a pattern of exposure and response observed in animals to humans are due to obvious differences, including differences in genetic makeup, size, life span, metabolism, absorption, and excretion of a substance.

EPA's estimates of lung cancer risks to the general population due to radon are based on human exposure-response data from epidemiologic studies of underground miners. Unlike many other environmental pollutants, therefore, special procedures for extrapolating from animals to humans (with their associated uncertainties) are not needed for radon. Further, although it is necessary to extrapolate from the risk observed in miners at occupational levels of exposure to estimate the risk from residential exposure, the extrapolation is not large. Average cumulative lifetime radon exposures to the general public are only slightly below those for which excess risk can be demonstrated in miners (about 40-70 WLM). For example, exposure to the annual average U.S. cumulative radon exposure of 0.242 WLM over a lifetime (74 years) is equivalent to a cumulative exposure of approximately 18 WLM, and exposure to a concentration of 4 pCi/L over a lifetime is equivalent to a cumulative exposure of approximately 57 WLM, assuming that the equilibrium factor is 0.5 and the fraction of time spent indoors is 75% (see Exhibit 2-13 for more comparisons of WLM and pCi/L based on these assumptions).

Assumption of Linear Exposure-Response Relationship

In assessing residential radon risk, EPA assumes that the exposure-response relationship is linear at low exposures and exposure rates. This assumption is consistent with the evidence for linearity at a wide range of cumulative exposures in the radon epidemiologic studies of underground miners. There is no evidence of a threshold for lung cancer response from radon exposure, that is, a level of radon exposure below which no increased risk of lung cancer would exist. It is generally recognized that even low doses of alpha radiation produce DNA damage that cannot always be repaired. Research further indicates that at low doses of alpha radiation the dose-response relationship for cell transformation and tumorigenesis is linear and independent of dose rate (NCRP 1980; NAS 1988).

The BEIR IV Committee has stated that "linear models are adequate for extrapolation to low doses of high linear energy transfer (LET) radiations" (NAS 1988). The Committee noted that the only significant evidence against linearity in their analysis was the observation of a decrease in risk per WLM at very high exposure levels (above 2,000 WLM), and these levels were excluded from their primary analysis (NAS 1988). Significant increases in lung cancer mortality have been observed in miners at a wide range of cumulative exposures, including low levels comparable to a lifetime residential exposure at 4 pCi/L. In developing their radon risk projection models, both the ICRP 50 and BEIR IV Committees assumed that the relationship between cumulative radon progeny exposure and lung cancer risk is linear at low exposures. The EPA's Science Advisory Board (SAB) recommended that the Agency continue to assume a linear exposure-response relationship when extrapolating risks from miners to estimate the risks from residential exposures (U.S. EPA 1988).

Risk Projection Models

1. History

Several models have been developed to characterize the lung cancer risk associated with exposure to radon daughters in the residential environment, based on studies of miners exposed to radon. EPA has relied primarily on relative risk projection models to estimate radon risks to the public. This section reviews the models that EPA used in the past to calculate radon risk estimates. Section 2 that follows describes EPA's current approach to the estimation of radon risks.

In general terms, the relative risk models assume that the rate of excess lung cancer due to radon increases with the baseline rate of lung cancer in the population due to all other causes. This means that the estimated number of deaths due to radon will vary depending on the baseline lung cancer mortality rate, which varies with age, sex, and other factors such as the prevalence of smoking in the population. Since smoking accounts for most of the baseline lung cancer risk, this approach is consistent with the assumption that radon and smoking act multiplicatively in causing lung cancer (Puskin and Yang 1988). A multiplicative relationship exists when the relative risk associated with combined exposure to both factors is approximately equal to the product of the relative risks associated with exposure to either of the factors alone.

Prior to 1988, EPA used a constant relative risk model to project radon-induced lung cancer risk in the general population. The model was of the form:

$$r(a)=r_0(a)\,[1 + \beta W^*] \quad (1)$$

where:

$r(a)$ represents the lung cancer mortality rate at age a from all causes including radon exposure,

$r_0(a)$ is the age-specific baseline lung cancer mortality rate,

β is the relative risk coefficient, and

W^* is the cumulative "exposure equivalent" up to age a-10.

EPA's SAB recommended assuming a range of 1 to 4%/WLM for β. The "exposure equivalent" was an adjustment to the cumulative exposure estimate (WLM) to account for age-specific differences in airway dimensions and surface area, respiratory frequency, and tidal volume. EPA assumed that the dose per WLM was 40% lower for residential exposures than for exposures received in underground mines.

The constant relative risk model was used in conjunction with a lifetable analysis governed by 1980 U.S. vital statistics, to project a risk factor of 460 fatal cancers/10^6 person-WLM with an uncertainty range of 230-920 fatal cancers/10^6 person-WLM. EPA's estimate of annual lung cancer deaths was 5,000-20,000 lung cancer deaths per year (U.S. EPA/Office of Radiation Programs 1987).

In May 1988, EPA reviewed its risk assessment methodology in response to the publication of the BEIR IV report (NAS 1988) and the issuance of a draft risk assessment for the National Emission Standards for Hazardous Air Pollutants (NESHAPS) rulemaking on radionuclides under the Clean Air Act (U.S. EPA/Office of Radiation Programs 1989). The SAB recommended that EPA use both the BEIR IV and the ICRP 50 reports to determine a range of radon risk estimates, and suggested using the average of the central values calculated with these two risk projection models to derive EPA's risk estimate (U.S. EPA 1988). The SAB did not recommend use of the NCRP 78 model, which is an absolute risk model. Absolute risk models have been described as less appropriate for the estimation of lifetime radon risk, as they do not assume a temporal correlation with the baseline lung cancer rate (ICRP 1987). Moreover, the NCRP model presumes that the effects of radon and cigarette smoke are additive, contrary to evidence from the Colorado Plateau Study. The SAB also recommended that EPA assume that indoor and mining exposures give the same dose per WLM of cumulative radon exposure. This was also the assumption used by the BEIR IV Committee. For assessing the risks due to childhood exposures, the SAB recommended that EPA rely on the ICRP 50 model. This model assumes a three-fold higher risk coefficient for exposures to people under 20 years of age. EPA used the BEIR IV and the ICRP 50 models in conjunction with a lifetable analysis and 1980 mortality data to estimate risk factors (lung cancer deaths/10^6 person-WLM) and annual deaths from lung cancer.

In deriving these rates with both models, EPA adjusted the age-specific baseline rates of lung cancer mortality, $r_o(a)$, by eliminating deaths due to average background radon exposure. Relative risk is normally used to determine the excess cases of a disease in an exposed population compared to an unexposed population. But in the case of radon, exposure is universal. The 1980 age-specific baseline rates of lung cancer include a radon contribution to lung cancer deaths. Therefore, EPA derived "radon free" background lung cancer rates, that is, reduced to reflect the removal of the age-specific radon contribution to these rates. EPA assumed that the average background radon exposure was 0.25 WLM/year. The net effect of this adjustment was to reduce the lifetime risk estimates given by the BEIR IV model by approximately 15% (U.S. EPA/Office of Radiation Programs 1989). In this way, the relative risk models do not project higher rates of excess radon-induced lung cancer based on lung cancer rates that already include lung cancers due to background radon. In their calculation of lifetime population risks from indoor radon exposure, the ICRP 50 Committee corrected the baseline lung cancer rate assuming an average exposure rate of 0.2 WLM/year (ICRP 1987). The BEIR IV Committee did not incorporate this correction in their risk estimates (NAS 1988). EPA adjusted both models assuming an average exposure rate of 0.25 WLM/year.

EPA's calculations using the adjusted BEIR IV model yielded a risk factor of 305 lung cancer deaths per 10^6 person-WLM, and an estimated 18,300 annual lung cancer deaths. The calculations using the adjusted ICRP 50 model yielded a risk factor of 420 lung cancer deaths per 10^6 person-WLM, and an estimated 25,200 annual lung cancer deaths. EPA then averaged the values calculated using each of these models separately. This gave a risk factor of 360 lung cancer deaths per 10^6 person-WLM and an estimated 21,600 annual lung cancer deaths.

The following sections discuss the ICRP 50 and the BEIR IV models and the adjustments made by EPA.

a. ICRP 50 Model

The ICRP 50 Committee developed their constant relative risk model based on the lung cancer incidence observed in three of the miner cohorts (U.S., Ontario, and Czechoslovakian uranium miners) and on information from the studies of the atomic bomb survivors (ICRP 1987). The model assumes a constant rate of excess lung cancer risk due to radon over time and age; that is, the increase in relative risk for a given exposure is constant over time after exposures end. The ICRP 50 model is written:

$$r(a) = r_o(a) + r_o(a)\beta W_{(a-10)} \qquad (2)$$

where:

$r(a)$ represents the rate of lung cancer mortality from all causes (including radon) at a specific attained age a,

$r_o(a)$ is the age-specific baseline lung cancer mortality rate (corrected for background radon exposure),

β is the relative risk coefficient for radon-induced lung cancer,

$W_{(a-10)}$ is the cumulative radon progeny exposure up to age a except for the previous 10 years. The 10 years represent the assumed minimum latency period between exposure and cancer development, and

$r_0(a)\beta W_{(a-10)}$ represents the age-specific rate of excess lung cancer risk due to radon.

The ICRP 50 model does not assume that the excess risk of lung cancer in miners due to radon can be directly transferred (extrapolated) to the general population. To adapt a relative risk model derived from studies of miners for use in estimating risks to the general population, the ICRP made certain adjustments. The radon relative risk coefficient from the miner data (β) was modified by the ICRP Committee to reflect presumed differences between mine and indoor radon exposures.

First, because of potential co-carcinogenic influences that may be present in the mines but not indoors (e.g., exposures to vapors, dust, and other forms of radiation), the ICRP assumed that the risk coefficient for indoor exposures would be only 80% of that from mine exposures.

Second, due to potential differences between indoor and mine exposure conditions, the ICRP Committee assumed that the absorbed dose of alpha radiation per unit of cumulative radon exposure (WLM) for the general population is only 80% of that for miners.

Third, based on findings from studies of the atomic bomb survivors, the ICRP Committee assumed a three-fold higher risk coefficient for exposures to people under 20 years of age. This had the effect of increasing overall lifetime risk by approximately 40% (Puskin and Nelson 1989).

When these correction factors are applied to the miner relative risk coefficient, the ICRP Committee's risk coefficients for indoor exposures to radon are:

β = 1.9% per WLM for exposures at ages $\leq$ 20 years
β = 0.64% per WLM for adult exposures > 20 years

When EPA used the ICRP 50 model to estimate lung cancer risks to the general population, a slight modification was made to these ICRP Committee risk coefficients (U.S. EPA/Office of Radiation Programs 1989). EPA eliminated the assumption that the absorbed dose of alpha radiation per unit of cumulative radon exposure (WLM) for the general population is only 80% of that for miners, and assumed instead that there was a 1:1 relationship between dose per unit exposure in mines and in homes. This was based upon a recommendation from the SAB and evidence considered by the BEIR IV Committee. The recalculated risk coefficients are then:

β = 2.4% per WLM for exposures at ages $\leq$ 20 years
β = 0.8% per WLM for adult exposures > 20 years.

b. <u>BEIR IV Model</u>

The BEIR IV Committee developed their relative risk projection model based on detailed analyses of four major cohort studies of underground miners (NAS 1988). These were the U.S., Ontario, and Saskatchewan (Beaverlodge) uranium miners and the Swedish (Malmberget) iron miners. The Committee reviewed both the original study data as well as unpublished material, and conducted a reassessment of the data using new statistical methods.

The Committee found a pattern of increased risk of lung cancer that was very consistent among the four groups. In the U.S. uranium miners, a decline in relative risk with age and with time since exposure was observed. There was no consistent effect on relative risk from age at first exposure or from rate of exposure. As additional data are collected from ongoing follow-up of the miner cohorts, the lifetime pattern of risk from radon exposure may be further clarified.

The BEIR IV modified relative risk model departs from a constant relative risk model (such as the earlier EPA model and the ICRP 50 model) by including additional assumptions about the effects of time since exposure (TSE) and of age. The BEIR IV preferred TSE model can be written as:

$$r(a) = r_0(a)[1 + \beta\gamma(a)\,(W_1 + 0.5W_2)] \tag{3}$$

where:

$r(a)$ is the lung cancer mortality at attained age a due to all causes including radon exposure,

$r_0(a)$ is the age-specific baseline lung cancer mortality rate,

$\gamma(a)$ is the age-specific adjustment to the relative risk coefficient for radon (β = 0.025), with

$$\gamma(a) \begin{cases} = 1.2 \text{ when } a < 55 \text{ years} \\ = 1.0 \text{ when } a \text{ is } 55\text{-}64 \text{ years} \\ = 0.4 \text{ when } a \geq 65 \text{ years.} \end{cases}$$

The $\gamma(a)$ adjustment decreases the radon-induced lung cancer risk with age. This incorporates the Committee's finding that excess relative risk in the miners decreased with age at risk.

$(W_1 + 0.5W_2)$ represents cumulative lifetime exposure up to age a modified as follows:

W_1 = cumulative exposure occurring from 5-15 years before age a, and

W_2 = cumulative exposure up to age a-15 years.

Since W_2 is reduced by 50%, the model gives less weight to exposures more distant in time since exposure. This reflects the Committee's conclusion that risk decreases with time since exposure as modeled for the four cohort studies of miners. Hence, the relative risk coefficient (β = 0.025) effectively varies from 0.5%/WLM to 3.0%/WLM, depending upon age at risk and time since exposure (Puskin and Nelson 1989). According to the model, $0.025\gamma(a)r_0(a)(W_1 + 0.5W_2)$ represents the rate of excess lung cancer due to radon.

2. Current Risk Projection Approach - Adjusted BEIR IV Model

In 1991, the Office of Radiation Programs requested that the SAB review proposed revisions to EPA's radon risk assessment methodology. The SAB recommended that the Agency use only the BEIR IV model for assessment of risk from residential exposure to radon. The recommendation to use only the BEIR IV model and discontinue use of the ICRP 50 model was based on several new pieces of information. The first was evidence from epidemiologic studies of a decrease in lung cancer risk with time since exposure, which had been incorporated into the BEIR IV model, but not the ICRP 50 model. The second was the publication of the BEIR V report (NAS 1990) and a study of Chinese miners exposed to radon gas (Lubin et al. 1990). These publications found no evidence of dependence on age at exposure for lung cancer. This was not consistent with the increased risk to children assumed in the ICRP 50 model (for further discussion of age dependence of risk see Section V).

EPA has made two adjustments to the BEIR IV model in estimating radon risks. The first modification was an adjustment of the age-specific baseline lung cancer mortality rate by eliminating deaths due to an average background exposure of 0.242 WLM/year, reducing the lifetime risk estimates by about 10%. The second modification was based on findings in the NAS' report, completed under a grant from EPA, *Comparative Dosimetry of Radon in Mines and Homes* (NAS 1991). The purpose of this report was to compare the dose to the bronchial epithelium per unit of radon progeny exposure in mines and homes, based upon a number of physical and biological factors which are expected to differ in the two environments. Among the factors considered in the report are the following: age, sex, aerosol size distribution, the unattached fraction of radon progeny, the breathing rate and route (oral vs. nasal), the pattern and efficiency of deposition of radon progeny, the solubility of radon progeny in mucus, and the growth of aerosols in the respiratory tract (for further discussion of this report see Section V.1.d.). This comparison of exposure-dose relations in the mining and home environments indicated that the dose per unit of exposure to radon progeny is approximately 30% lower in the home environment. This finding was expressed in terms of a K-factor (K = 0.7).

Risk is presumed to be proportional to the dose received by target cells in the bronchial epithelium. Therefore, in calculating the risks for residential exposures, it is appropriate to multiply the risk coefficient in the BEIR IV model by a factor of K.

Taking K = 0.7 for both sexes and all ages, the modified BEIR IV model becomes:

$$r(a) = r_0(a)[1 + 0.0175\,\gamma(a)\,(W_1 + 0.5W_2)] \qquad (4)$$

where the parameters are as described in equation 3, with the difference that $\beta = 0.0175$ as a result of the adjustment for the factor $K = 0.7$.

The most recent estimates of the deaths/person-WLM and annual U.S. lung cancer deaths calculated using the BEIR IV model are discussed in Section V of this chapter.

3. Lifetable Derivation: Rate of Radon-Induced Excess Lung Cancer Death

Because data are not available on the actual incidence rate of lung cancer death due to radon for the general population, the risk projection models are used to calculate a rate. (Although death certificates may identify cause of death, it is not possible to differentiate between the lung cancer due to radon and that due to smoking, or any other cause.)

To calculate a lifetime, age-averaged rate for radon-induced lung cancer in the general population, EPA has used a standard lifetable analysis in conjunction with the BEIR IV model as adjusted by EPA (U.S. EPA/Office of Radiation Programs 1989). The lifetable approach was used to account for the effect of competing causes of death. For each year of the lifetable calculation, the rate of age-specific death due to all causes competes with the rate of age-specific radon-induced lung cancer death. This is important because of the inherent delay between radon exposure and the occurrence of lung cancer. During and after the period of radon exposure and before cancer appears, a potential lung cancer victim is exposed to other causes of death (e.g., auto accidents, cardiovascular disease). When people die from these other causes, they are no longer in the "pool" of people at risk of dying from lung cancer due to radon.

EPA uses a lifetable for a general population that reflects the 1980 U.S. age-specific mortality rates and vital statistics. The lifetable starts out with the "birth" of 100,000 hypothetical persons who are followed from birth to death, with no member living longer than 110 years. Therefore, in each year of the lifetable, all members of the "birth cohort" (minus those that have died from various causes in the previous year, including radon-induced lung cancer) are the same age.

Using the BEIR IV model, the lung cancer deaths due to a given cumulative radon exposure (person-WLM) can be calculated for each year of the lifetable over a full life span, from birth to 110 years. The deaths for each year are calculated based on the cumulative radon exposures for that year and for the preceding years as determined by the model. The age-specific rate of lung cancer death due to radon derived with the BEIR IV model is $0.0175\gamma(a)r_0(a)(W_1 + .5W_2)$ (see previous discussion of the BEIR IV model).

In deriving the risk per unit exposure, EPA assumes each member of the lifetable receives a constant low level of radon exposure (0.001 WLM/year) from birth to death. (A constant low level of exposure over 110 years is used because the lifetime risk estimate would no longer increase linearly if there were high cumulative radon exposures.) Thus, from the beginning of the lifetable, 100,000 persons are subjected year after year to the age-specific mortality rates of the 1980 U.S. population and to an increased age-specific lung cancer mortality rate due to low level radon exposure experienced up to that year.

As each age-specific radon-induced lung cancer mortality rate is calculated (e.g., for age 10-11), it is applied to the surviving lifetable population for that year. This generates the excess deaths that year for the cumulative radon exposure experienced by the lifetable population up to that year (i.e., Y excess lung cancer deaths per Z person-WLM). These deaths are then reflected in the overall mortality rate from all causes for the next year (e.g., for age 11-12). Therefore, due to deaths from all causes, each year there are fewer and fewer of the original 100,000 members of the lifetable population still alive and at risk from radon exposure. The lung cancer mortality for cumulative radon exposure is recalculated each year of the lifetable based on the revised population and cumulative radon exposure for that next year.

After this has been done for each of the 110 years in the lifetable, the excess radon-induced lung cancer deaths for each year can be summed to give a total lifetime number of deaths for the population. The collective exposures to the cohort from each year (i.e., over all ages) are also summed to give the lifetime population exposure.

4. Applying the Rate of Radon-Induced Lung Cancer to the U.S. Population

EPA's estimation of annual deaths due to radon (see Section V) is based on a steady-state population, i.e., one that does not change from year to year. The steady-state population is constructed to have people of all the same age groups, and in the same proportion, as those in each year of the lifetable cohort (i.e., covering newborns to 110 year-olds). For each year of exposure, the population will have the same proportion of exposure (person-WLM) by age as a birth cohort receiving a constant lifetime exposure. This is essentially the population that would result from a constant birth rate of 100,000 per year (birth cohort) with the age-specific mortality rates of each cohort defined by the 1980 mortality rates and vital statistics. Therefore, the number of radon-induced lung cancer deaths per person-WLM for constant lifetime exposure to the single birth cohort is the same as that for an annual exposure to all members (at various ages) of the stationary population (U.S. EPA/Office of Radiation Programs 1989).

Because these estimates of annual lung cancer deaths due to radon were derived using a lifetable defined by 1980 statistics, they are specific to the 1980 population. Since a relative risk approach is used, the estimated number of excess lung cancer deaths due to radon exposure will change as these background rates of lung cancer mortality change. As new mortality rates and vital statistics become available, EPA will revise its estimates of radon risk.

Section V: Risk Characterization

EPA's characterization of the risk associated with residential exposure to radon has two primary components: calculation of the numerical estimates of risk, and presentation of the information needed to judge the overall significance of the risk estimates.

In addition to deriving estimates of risk from low-level residential radon exposures and the number of lung cancers induced by radon each year using the BEIR IV model, EPA has identified and quantified, to the extent possible, the major sources of uncertainty in these estimates and has made a quantitative estimate of overall uncertainty. This includes a discussion of the uncertainty associated with both the risk factor and the estimate of average residential radon exposure. Since the uncertainty analysis is critical to the interpretation of the risk estimates, this analysis is presented first, followed by EPA's current estimates of risk.

Uncertainty Issues

In the past, EPA has quantified uncertainty using the ICRP uncertainty range which represents a range of risk coefficients from the miner studies (ICRP 1987). The Agency's current uncertainty analysis is based on the methodology used in the National Institutes of Health report on the development of radioepidemiological tables (NIH 1985). According to this approach, each component of uncertainty is treated as independently affecting risk by a multiplicative factor, and the upper and lower bounds for each component of uncertainty are treated as the limits of a 90% "confidence interval." These are referred to as "credibility intervals" by NIH to emphasize the element of subjectivity involved in their estimation. In estimating the overall uncertainty, each component is further assumed to be distributed lognormally on each side of the nominal geometric mean (G) of the upper and lower bounds. The lower and upper bounds are then defined by multiplicative factors L and U, such that the limits of a 90% confidence interval are given by LxG and UxG, respectively. The selection of a lognormal distribution is somewhat arbitrary, and much of the justification is one of practicability and mathematical tractability.

Sources of uncertainty that do not readily lend themselves to quantification were not included in the uncertainty analysis, but were addressed qualitatively.

1. Risk Factor and Associated Uncertainty

The following are sources of uncertainty associated with the estimate of lifetime risk that were addressed in the uncertainty analysis: (a) statistical variability in the miner data, (b) projection of risk beyond period of epidemiologic follow-up (projection of risk over time), (c) age dependence of risk, (d) extrapolation from mines to homes, (e) influence of mine exposures other than radon, (f) exposure-rate effect, (g) extrapolation to females, and (h) relationship between radon risk and smoking.

Of the eight sources of uncertainty that were addressed, four were included in the quantitative analysis. These were: variability in the miner data, time projection, age sensitivity, and extrapolation from mines to homes based on bronchial dose.

a. Statistical Variability in the Miner Data

As discussed in the BEIR IV Report, the statistical variability in the miner data is most readily addressed in terms of a constant relative risk model. It is recognized that this is not the model preferred by the BEIR IV Committee for describing the temporal and age dependence of the risk; however, the uncertainty in the time and age projections can be treated separately.

The BEIR IV Committee analyzed four cohorts of miners. Exhibit 2-3 shows the risk coefficient derived from each study, assuming a constant relative risk model, and, in parentheses, the respective multiplicative standard error (MSE).

EXHIBIT 2-3
BEIR IV ESTIMATES OF RISK COEFFICIENTS FOR A CONSTANT RELATIVE RISK MODEL[a]

Cohort	% Increase in Relative Risk/WLM β (MSE)
Eldorado (Beaverlodge)	2.6 (1.5)
Ontario	1.4 (1.6)
Malmberget	1.4 (2.6)
Colorado Plateau	0.6 (1.5)

[a] Source: NAS 1988.

The values in Exhibit 2-3 are based on an internal analysis in which the miner cohort effectively served as its own control (NAS 1988). Results from an external analysis of each cohort differ only slightly from these values. Treating the errors as lognormally distributed, the combined estimate of the risk coefficient and standard error from the four studies is calculated to be:

$$\ln \beta = \frac{\sum_i(\ln \beta_i/\ln^2 S_i)}{\sum_i(1/\ln^2 S_i)} \tag{5}$$

$$1/(\ln^2 S) = \sum_i(1/\ln^2 S_i)$$

where:

the β_i values are the respective risk coefficients estimated for the individual studies, and

the S_i values are the corresponding multiplicative standard errors.

Substituting from Exhibit 2-3 above, the combined estimate for β is 1.30%/WLM (S = 1.27). This compares with the BEIR IV combined estimate of 1.34%/WLM and an MSE of about 1.3. The small difference between the estimates given in BEIR IV and those given here (calculated by EPA) may reflect rounding errors or a difference in method for arriving at the combined estimate (the BEIR IV report does not describe its calculation).

As noted in BEIR IV, agreement among the studies is quite good and, on statistical grounds, there is no reason to reject any of the four studies as an outlier. Further support for the reliability of the estimate is provided by other studies not included in the BEIR IV analysis. A noteworthy example is a recent analysis of the Czechoslovakian uranium miner data that derived the estimate of 1.5%/WLM (Sevc et al. 1988). In addition, based on an analysis of 8 miner cohorts, Lubin (1988) obtained a combined estimate for β of 1.5%/WLM with an MSE of 1.2. It was noted that none of the four studies considered by BEIR IV can be regarded as a statistical outlier. It is nonetheless instructive to examine the sensitivity of the combined estimate for β to possible bias in a single study. If, on the one hand, the cohort yielding the highest estimate (Eldorado) is removed from the analysis, the estimate of β is reduced from 1.30 to 0.90%/WLM. On the other hand, removing the Colorado cohort, which yields the lowest estimate, increases the combined estimate to 1.93%/WLM. Thus, the risk factor is fairly robust with respect to potential systematic error in any one study.

Upper and lower confidence limits on β associated with the statistical variability in the miner data are readily obtained from the estimated MSE of 1.3: $U = L^{-1} \approx (1.3)^{1.64} \approx 1.5$. That is, considering only statistical variability in the miner data, the 90% confidence interval for β is (0.013/1.5 - 0.013 $\times$ 1.5), or about 0.0087 to 0.0195.

It is important to emphasize that this estimate of uncertainty in the miner data does not include uncertainty associated with systematic errors in the assignment of exposure to individual miners, and the possible influence of mine exposures other than radon.

b. <u>Projection of Risk Over Time</u>

When data from all four of the BEIR IV miner cohorts were combined, the resulting risk coefficient was 1.3%/WLM. This value represented a best estimate for the limited period of follow-up in the cohorts, assuming a constant relative risk model and a 5-yr minimum latency. Further statistical analysis, however, revealed an apparent decrease in the relative risk with (1) age at risk and (2) time since exposure. Consequently, the BEIR IV Committee preferred model incorporates age-at-risk and time-since-exposure dependent parameters. The net effect of these model changes was to reduce the estimated lifetime risk due to lifetime exposure by about one-third. In the preferred model, the risk coefficient effectively varies from 0.5 to 3%/WLM, and the lifetime risk estimate is about the same as would be projected using a constant relative risk model with β = 0.9%/WLM.

Although there is statistically significant evidence of a temporal fall-off in risk in the data for four cohorts, much uncertainty remains as to the rate and eventual degree of the fall-off. Some of the observed decline may be due, at least in part, to changes in miner smoking habits over time or other unknown confounding factors. However, the BEIR IV Committee concluded that it was unlikely that the decrease could be accounted for simply by changes in the smoking habits of the miner cohorts. Additional follow-up on temporal changes in the smoking habits of the miners may help settle this issue.

To obtain a reasonable bound on the possible underestimate in risk due to uncertainty over the time projection, a constant relative risk model was assumed. As indicated above, this assumption increased the risk estimate by about a factor of U = (1.3%/WLM)/(0.9%/WLM) ≈ 1.5.

An estimate of the corresponding lower bound was obtained using an approach suggested in the BEIR IV Report (NAS 1988). Although the preferred model assumes an excess risk projection over a full lifetime, useful follow-up on the miners extends only for 30-40 y after their first exposure. To obtain a reasonable lower bound estimate, therefore, it was assumed that the excess risk ends completely about 40 y subsequent to exposure. As pointed out in BEIR IV, this is equivalent to stipulating that W_2, in the preferred model, includes only exposure accrued 15-40 y before the age at risk. Quantitatively, the effect of such a postulated cut-off would be to reduce the general population risk estimate by about 33%, i.e., by a factor of L = 1/1.5 (see Exhibit 2-4).

As shown in Exhibit 2-4, most of the effect of such a cut-off is associated with childhood exposures. The estimated risk due to exposures at age 0 and 10 is reduced by a factor of 100 and 10, respectively. For all exposures between ages 0 and 20, the average risk is decreased by a factor of 7.5; by comparison, for all exposures above age 20, the average risk is decreased by only 10%.

EXHIBIT 2-4
AVERAGE LUNG CANCER MORTALITY RISK PER 10^6 WLM BY EXPOSURE GROUP, WITH AND WITHOUT 40-YR CUTOFF IN EXPRESSION OF RISK[a/]

Projection Limit	Exposure Group by Age (y)				
	0	10	0-20	+20	All
None	322	328	328	294	304
40 Yr	3.4	32	44	264	205

[a/] Source: Calculated using the BEIR IV model (uncorrected for difference between dosimetry in homes and mines). The baseline lung cancer rate was corrected for the contribution of background radon exposure, which was assumed to be 0.25 WLM/y for this calculation.

c. Age Dependence of Risk

The BEIR IV Committee "did not find an effect of age at first exposure, after controlling for other correlates of age" (NAS 1988). However, their study population was comprised entirely of adults. Direct evidence on the sensitivity of children to radon is very limited. The only such information currently available is from a study of Chinese tin miners that included some individuals who started mining before age 13 (Lubin et al. 1990). No enhancement in the risk per unit exposure was seen in these individuals as compared to the rest of the cohort. However, given the sparseness of the data and the lack of control for smoking, no definite conclusions can be drawn from this study.

Atomic bomb survivors irradiated before age 20 show a highly statistically significant greater relative risk for radiogenic induction of cancer than those irradiated after age 20. Also, for lung cancer, a higher relative risk is observed for those irradiated before age 20, but the increase in risk is not statistically significant (Shimizu et al. 1988). Guided by the results obtained on the atomic bomb survivors (through 1978), the authors of the ICRP 50 report on radon risk assigned a 3 times higher relative risk coefficient to exposures received before age 20 (ICRP 1987).

The data on atomic bomb survivors are suggestive of a higher risk of radiogenic lung cancer in children, but epidemiologic follow-up is still too short for those exposed as children to express many lung cancers; consequently, no firm conclusions can be drawn. Additional follow-up should settle the question, but the applicability of the findings to the case of chronic radon daughter exposures would remain an issue.

As an upper limit on the possible effect of age at exposure on the BEIR IV risk estimate, the lifetime risk for a constant exposure rate was calculated using the preferred BEIR IV model revised to incorporate a 3 times higher risk coefficient for all exposures before age 20. The net effect of the modification was to increase the general population risk by about 60%. Accordingly, for this source of uncertainty, U is taken to be 1.6.

While there is no basis for postulating that the lungs of children are inherently less sensitive to the carcinogenic effects of radon daughters, a temporal fall-off in risk could effectively make the lifetime risk from childhood exposures very small. This possibility has already been addressed in the previous section and incorporated into the uncertainty relating to the temporal projection. Thus, with respect to age sensitivity, L is taken to be 1.0.

d. Extrapolation from Mines to Homes

As discussed previously, the risk coefficients used in the BEIR IV risk model were adjusted by a factor K = 0.7 to correct for an estimated lower bronchial dose per WLM in homes as compared to mines, based on information contained in the recent report, *Comparative Dosimetry of Radon in Mines and Homes* (NAS 1991). The NAS' report did not attempt to quantify the overall uncertainty in K, but it did indicate how much change would occur under some alternative sets of assumptions. That information was used here to develop an estimate of the uncertainty in K.

It is important, in this connection, to distinguish uncertainty from variability. The value of K appropriate to an individual will vary widely across the population, depending on personal characteristics (e.g., age) and on conditions in the home. For example, Table 3-4 of the NAS' report shows large standard deviations on K (typically, about 30%). In large part, these reflect calculated changes in K due to variability in the aerosol conditions that people are exposed to, not uncertainties in the value of K most characteristic of average exposure conditions.

Numerous assumptions have to be made in calculating the dose to target cells in the lung from a given radon exposure, and each of these assumptions is associated with some uncertainty. For example, the NAS' Committee considered various alternative classes of target cells: primarily, secretory vs. basal cells, and the whole bronchial tree vs. the upper (lobar/segmental) bronchi only. No preference was expressed regarding the type of cell; estimates were generally given for secretory and basal types separately or as the mean of the two. Assuming that secretory or basal cells alone are the target cells would increase or decrease the estimate of K by 6%, respectively, relative to the mean. The preferred estimates were based on target cells being distributed over the whole bronchial tree; if only the upper bronchi were to be considered, K would have been decreased by 5%.

Much of the uncertainty in K pertains to the dose contributed by unattached radon decay products in the home. According to estimates endorsed by the NAS' Committee, it was considered that about 2/3 of the total dose in homes comes from the unattached fraction. However, subsequent to the publication of new dosimetric information, the Committee presented revised estimates based on a modified set of assumptions. The modifications included a higher nasal deposition of unattached daughters and a smaller particle size in some mine locations. With the higher nasal deposition, the contribution of the unattached fraction in homes fell to roughly 50% (James 1991), and the estimate of K for each age group was reduced by about 20%.

The fraction (f_p) of radon daughter alpha-energy which is "unattached" (i.e., present as an ultrafine fraction) varies considerably between houses and, over time, even within a single house. The NAS' report adopted a value of 8% as typical for the indoor environment, and cited a range of 7% to 10% for indoor measured values in the absence of smoking. However, given differences in heating and cooking methods, air cleaning equipment, ventilation, etc., this range would seem to be too narrow. In this connection, the report noted that f_p may exceed 16% in houses with low aerosols.

Tobacco smoke provides additional attachment sites for radon decay products; consequently, f_p is expected to be lower in houses with a smoker present. The NAS' Committee assumed that, on average, f_p is reduced from 8% to 3% during waking hours in houses with a smoker, and to 1% during actual smoking. As a consequence, for waking hours, K is reduced by about 40% in houses with a smoker present. Taking into account sleeping hours, the reduction in average K is roughly 20%. The reduction would be less under the assumptions discussed above (higher nasal deposition of unattached daughters and smaller particle size in mines), since the contribution to dose from the unattached fraction would be less. Another aspect that should be considered in this connection is the higher equilibrium factor (defined in Section V.2.b.) expected in homes with tobacco smoke (Vanmarcke et al. 1988). Thus, <u>for a given radon concentration</u>, smoking will increase radon daughter concentration but decrease the unattached fraction. These changes will have opposite effects on dose and it is unclear whether the average dose, at a given radon concentration, would actually be higher or lower in homes with a smoker.

A critical assumption made by the NAS' Committee, which substantially reduced their estimate of K, pertains to the hygroscopic growth (by taking up and retaining moisture) of aerosols inside the respiratory tract. Except for certain nonhygroscopic aerosols produced by cooking and vacuuming, the Committee assumed a doubling of the aerosol particle size in the respiratory tract. Due to differences in the initial size distribution of the aerosols, assuming hygroscopic growth led to a reduction in calculated dose for homes but an increase for mines, especially during heavy exercise.

With no hygroscopic growth, K would be increased by about a factor of 1.6 (1.5, with modified assumptions on nasal deposition and particle size).

Summarized in Exhibit 2-5 are the average adult values calculated for K by the NAS' panel, with and without the modifications in nasal deposition and particle size, with and without growth (doubling) of aerosol size in the respiratory tract (NAS 1991). For children, the values of K would typically be increased by about 10%.

EXHIBIT 2-5
AVERAGE ADULT VALUES FOR K ADJUSTED FOR NASAL DEPOSITION AND HYGROSCOPIC GROWTH[a/]

Assumptions on Nasal Deposition and Particle Size	Growth	No Growth
Unmodified	0.70	1.1
Modified	0.58	0.88

[a/] Source: NAS 1991.

Another source of uncertainty in K is associated with uncertainty over the activity-weighted size distributions of radon progeny, both in mines and in homes. However, this uncertainty is not quantified in the NAS' Radon Dosimetry report.

In view of the recent data indicating higher nasal deposition of the unattached fraction and the expectation of at least some hygroscopic growth of particles in the respiratory tract, it would appear that the dose/WLM is, on average, higher in mines than in homes (i.e., $K < 1$). Taking into account possible enhanced nasal deposition of unattached daughters in conjunction with an assumed size doubling of hygroscopic particles, as well as some allowance for additional sources of uncertainty inherent in the NAS' analysis, $K = 0.5$ is assumed to be a reasonable lower bound. These conclusions must be regarded as tentative and subject to change in light of better information on aerosol conditions in mines and homes. Accordingly, a symmetric multiplicative uncertainty range $U = L^{-1} = 1.4$ is taken around the central value of $K = 0.70$.

e. <u>Other Mine Exposures</u>

Uncertainty associated with the presence of agents such as ore dust, arsenic, fluorides, silica and diesel fumes in the mining environment has been considered but not quantified in this analysis.

It has been suggested that these other mine exposures may have enhanced the risk of lung cancer in the miners by increasing cell turnover (NAS 1991). It is important to note, however, that the potential effects of other mine exposures have been taken into consideration in many of the miner studies. Sevc et al. (1984) reported that in the Czech uranium mines, a comparison of areas with similar radon levels but different arsenic levels showed no variation in lung cancer incidence.

Archer et al. (1985) conducted an analysis of five mining or milling groups and found that lung cancer rates correlated with radon exposure regardless of silica dust levels. Animal studies have shown that exposure to ore dust or diesel fumes simultaneously with radon does not increase the incidence of tumors produced by radon daughter exposures (Cross et al. 1991). The miner studies were conducted in different types of mines where environmental pollutants other than radon were present to varying degrees, and radon was the common exposure. The similarity in the estimates of lung cancer risk per WLM of radon exposure provides evidence of the role of radon as the primary carcinogen.

Further analysis of the potential effect of other exposures on lung cancer risk in the underground miners will be undertaken in the previously described joint analysis of 11 miner cohorts (Lubin 1991), which proposes to critically evaluate the variation of the exposure-response relationship with age, time after exposure, cigarette smoking, and other mine exposures.

f. Exposure-Rate Effect

Exposure levels in homes are generally lower than those the miner cohorts experienced. Laboratory studies indicate that the risk per unit dose from alpha irradiation, in general, and from radon daughters in the lung, specifically, is maximal at low dose rates. The same trend has been found in human studies (NAS 1988, Sevc et al. 1988, Howe et al. 1986). If risk decreases with increasing exposure rate, then the estimate of the risk coefficient derived from miner studies may in some cases be biased low relative to what is appropriate for use at the lower exposure rates in residences. In particular, it often has been suggested that the high exposure rates prevailing in the Colorado mines may be responsible for the comparatively low risk coefficient derived from the Colorado Plateau cohort study. A quantitative estimate of possible bias in the BEIR IV analysis associated with exposure rate might then be obtained by excluding the Colorado Plateau cohort from the analysis. As discussed in the previous section, this exclusion would increase the combined estimate of β by about a factor of 1.5, i.e., from 1.30%/WLM to 1.93%/WLM.

g. Extrapolation to Females

The BEIR IV Committee assumed that the same age-specific relative risks found for men could be applied to women. However, U.S. male lung cancer rates are substantially higher than female rates. As a result, the BEIR IV model projects a much higher risk of lung cancer attributable to radon for men than for women. There is, however, no evidence that males are inherently more susceptible to the disease; rather, most of the difference in baseline rates seems to be explained by differences in smoking habits. It follows that a great deal of the uncertainty regarding dependence of radon risk on gender revolves around the question of the radon-smoking interaction.

h. Relationship Between Radon Risk and Smoking

Issues surrounding the possible form of the radon-smoking interaction and its possible effect on radon risk are complex and rather intractable given the limited information available. No attempt is made here to quantify the uncertainty due to this source, but rather to outline what appear to be the major problems.

Information on the interaction between smoking and ionizing radiation in causing lung cancer is somewhat conflicting. Data on the atomic bomb survivors suggest additivity but are also consistent with a supra-additive or even a multiplicative interaction (Prentice et al. 1983). According to BEIR

V, despite lower smoking exposures, women have about the same excess risk of lung cancer attributable to radiation as men; thus, women appear to have higher relative risks.

The most important data currently available on the interaction between radon and smoking come from the Colorado Plateau miners, for which there is individual smoking information on all miners (changes in smoking habits in the cohort subsequent to 1968 have not yet been reported for the entire cohort however). Statistical analyses of this cohort yield results that are consistent with a multiplicative but not an additive model; however, a wide range of supra-additive and even supra-multiplicative models also give acceptable fits (NAS 1988). Other human data are rather limited, including support for everything from a multiplicative to a protective effect of smoking on radon risk (Lubin 1988, L'Abbé et al. 1991, Samet et al. 1991b, Sevc et al. 1988, Radford and St. Clair Renard, 1984). Animal studies, too, are conflicting: one study showing a synergism with tobacco smoke (Chameaud et al. 1980); another, a protective effect (Cross et al. 1982).

The EPA model of radon risk is based on the BEIR IV assumption that radon and smoking act multiplicatively in causing lung cancer. According to this model, the radon risk is proportional to the baseline lung cancer rate, so that as the baseline rate varies over time (due largely to changes in smoking habits), the projected rate of radon-induced lung cancers varies in parallel. Therefore, neglecting possible changes in average exposure rates, it is, according to the model, the proportion of all lung cancers attributable to radon that remains constant over time, not the absolute rate of radon-induced lung cancers. Thus, for past years when lung cancer rates were much lower than they have been recently, the model also projects much lower rates of radon-induced lung cancers.

An important element of uncertainty concerns the temporal dependence of the smoking-radon interaction. Young children are not generally smokers, but some of them will smoke when they get older. By employing a relative risk model in which the risk of lung cancer due to childhood exposure increases with the baseline rate at the age at risk, EPA effectively assumes that the risks from childhood radon exposure and adult smoking multiply. This may not be unreasonable since animal data suggest that tobacco smoke exposure acts synergistically with prior radon exposure in causing lung cancer (Chameaud et al. 1980). Moreover, such an assumption is consistent with the widely held view that radiation is primarily an initiator of carcinogenesis while tobacco smoke acts both as an early and late stage carcinogen (Doll 1978). Another issue concerns risks to former smokers, either due to radon exposures received prior to cessation of smoking or afterward. Additional follow-up on the U.S. miners may shed light on this question.

Finally, it should be noted that the risks calculated here are based on 1980 lung cancer mortality rates. These rates are continually changing in response to changes in smoking patterns. This limits the ability of the current risk estimates to predict future radon-induced lung cancers.

i. Overall Uncertainty in the Risk Factor

Summarized in Exhibit 2-6 are the upper and lower multiplicative uncertainties due to the sources of error that could be estimated reasonably. With regard to the epidemiologic studies of miner cohorts, there is additional uncertainty not addressed in this analysis associated with systematic errors in the assignment of exposures to individual miners based on area radon and radon decay product monitoring and, in some cases, highly questionable extrapolations over space and time. Also not included is the possible effect of other mine exposures. In extrapolating the miner data to the situation of radon in homes, additional uncertainty exists regarding the dependence of radon risk on exposure rate, gender, and smoking. These also are not addressed quantitatively here.

EXHIBIT 2-6
QUANTITATIVE ESTIMATES OF UNCERTAINTIES IN RADON RISK[a/]

Source of Uncertainty	U_i	$(L_i)^{-1}$
Statistical Variability in Miner Data	1.5	1.5
Projection of Risk Over Time	1.5	1.5
Age Dependence of Risk	1.6	1.0
Extrapolation from Mines to Homes	1.4	1.4

[a/] Source: Calculated in text.

Following, in outline, the calculations presented in the National Institutes of Health report on the development of radioepidemiological tables (NIH 1985), each source of uncertainty (i) is characterized by a lognormal probability distribution where the geometric mean $G_i = (U_i \times L_i)^{1/2}$ and where the geometric standard deviation (GSD) S_i is given by $U_i/L_i = S_i^{3.29}$. The latter equation is equivalent to the assumption that the 90% "confidence interval" of the respective distribution is $(G_i \times L_i, G_i \times U_i)$. The combined uncertainty distribution due to all four sources is then lognormal with a geometric mean G given by

$$\ln G = \ln G_i + \ln G_2 + \ln G_3 + \ln G_4 \tag{6}$$

and a GSD given by

$$\ln^2 S = \ln^2 S_1 + \ln^2 S_2 + \ln^2 S_3 + \ln^2 S_4 \tag{7}$$

Substituting from Exhibit 2-6 then, G = 1.26(5). The 90% "confidence interval" encompassing all four sources of uncertainty is then $(G \times L, G \times U)$ where $U^2 = 1/L^2 = S^{3.29}$. Solving, U = 1/L = 2.02.

The four components of uncertainty were defined relative to a central estimate of risk based on the BEIR IV model, adjusted to correct for an assumed 30% lower dose per WLM in homes compared to mines. Upper and lower bound estimates that reflect the combined uncertainty are then obtained by multiplying the central estimate by $G \times U = 2.56$ and $G/U = 0.625$, respectively. Thus, according to this analysis, the actual risk could be about 2.6 times higher than estimated by the adjusted BEIR IV model or about 1.6 times lower.

The adjusted BEIR IV model, used in a lifetable calculation in conjunction with U.S. 1980 vital statistics, yields a risk factor of 224 lung cancer deaths per 10^6 person-WLM, for constant lifetime exposure. Multiplying this value by G, the geometric mean risk estimate is 283 lung cancer deaths per 10^6 person-WLM. The estimated uncertainty range is then found to be 140 to 570 lung cancer deaths per 10^6 person-WLM.

As discussed previously, this range does not include several potentially important sources of uncertainty, such as errors in miner dosimetry and the effects of exposure rate, gender, and smoking, which cannot be quantified at present. In general, these uncertainties could produce underestimates or overestimates of the risk. Also, it should be emphasized that the uncertainties in risk to specific subpopulations — including children, never smokers, and former smokers — are considerably higher.

2. Average Residential Radon Exposure and Associated Uncertainty

The average residential exposure is estimated to be:

$$E = C\,[F \times 0.01\,\mathrm{WL/(pCi/L)}][\,\Omega \times 51.6\ \mathrm{WLM/WL\text{-}y}] \tag{8}$$

where C is the average radon concentration in homes, F is the average equilibrium factor, and Ω is the average occupancy factor. Based on a review of available information, EPA has determined that the previously employed values of $F = 0.5$ and $\Omega = 0.75$ are still appropriate for estimating exposure and risk, but the estimate of C is revised slightly downward from 1.29 to 1.25 pCi/L (a discussion of the estimates for these parameters and their uncertainties is provided below). Substituting into the equation above, the estimated average exposure is 0.242 WLM/y.

a. Estimate of Average Radon Concentration

EPA's National Residential Radon Survey (NRRS) of U.S. residences estimates that the annual average radon concentration to which people are exposed in their homes is about 1.25 pCi/L. This value reflects an average over all frequently occupied areas for each home, averaged over all homes. The standard error in the measurement is 0.07, implying a 90% statistical confidence interval of about 1.14 to 1.36 pCi/L.

The radon level in each house, as determined by the NRRS, represents year-long measurements taken at one or two locations on each floor defined as a frequently occupied area. Frequently occupied areas are defined as any level of the home, including basements, which contain a family room, living room, playroom, den, or bedroom. Also included as frequently occupied areas are basements which may not contain one of the previously mentioned room types, but are occupied by a resident for more than four hours per day, such as a frequently utilized workshop. The annual average radon concentration estimate of 1.25 pCi/L does not reflect the time spent on floors not classified as frequently occupied ares. Occupancy data, which reflect the total time each individual spent in the house and on each floor of that house, were collected in the NRRS, but they primarily reflect summertime activity patterns, which are likely to be atypical; consequently, they were not used to adjust the estimate of the average indoor radon concentration. Had floor occupancy data been used to weight the floor measurements, the estimate of the annual average radon concentration would have been diminished only slightly — by about 6%.

In order to allow for these additional uncertainties discussed in the preceding paragraph, an estimate is adopted of the overall uncertainty in the average radon concentration to which people are exposed in their homes, a range of 1.1 to 1.4 pCi/L, which is slightly wider than the statistical 90% confidence interval from the NRRS cited above.

b. Estimate of Average Equilibrium Factor (F)

The equilibrium factor (F) is defined as the ratio of the potential alpha energy concentration in the existing mixture to that which would exist if all short-lived daughters were in equilibrium with the radon present (NCRP 1988). Measured values of F are highly variable between houses (or even within a single house), depending on ventilation rates, aerosol concentrations, and other factors. The equilibrium factor rises with increasing aerosol concentration because the radon decay products attach more rapidly, reducing plate-out onto walls and other surfaces. Consequently, F will be increased in the presence of a smoker. From a risk standpoint, the situation is less clear, however, since concomitant with the increase in F will be a decrease in the unattached fraction, f_p; the changes in F and f_p are predicted to have opposite effects on lung dose.

Based primarily on measurements in 21 houses reported by George and Breslin (1980), EPA has used a value of 0.5 for F. The NAS' radon dosimetry report suggests that the average value is more likely to lie between 0.3 and 0.4, but that conclusion seems to rest on measurements carried out in a relatively small number of houses — mostly European. Recently, two large surveys of houses in the U.S. have been completed. The first was a survey of 200 houses conducted by the State of New Jersey that gave a mean equilibrium factor of 0.45 (NJ Department of Environmental Protection 1989). The second, a multi-state survey of 113 homes conducted by Radonics, yielded an average value of 0.51 (Radonics 1991). The data from the latter were found to be approximately lognormally distributed; the geometric and arithmetic means of the distribution were 0.54 and 0.56, respectively.

Exhibit 2-7 summarizes various measurements of F. For the most part, they represent "grab sample" measurements and/or measurements taken in a small number of houses. Only the Radonics study (1991) seems to reflect measurements taken over several days in a large number of houses. In

EXHIBIT 2-7
MEASURED VALUES OF THE EQUILIBRIUM FRACTION, F, IN HOMES

Location	F	Reference
Austria	0.60	Steinhausler et al. 1980
Finland	0.47	Makelainen 1980
Norway	0.50	Stranden et al. 1979
Sweden	0.44	Swedjemark 1983
nonsmoker	0.46	Jonassen and Jensen 1989
smoker	0.51	Jonassen and Jensen 1989
United States		
cellars	0.52	George and Breslin 1980
living areas	0.63	George and Breslin 1980
	0.33	Israeli 1985
	0.45	NJ Dept. of Environ. Protection 1989
	0.51	Radonics, Inc. 1991
West Germany	0.37	Wicke and Porstendorfer 1982
	0.34	Keller and Folkerts 1984
no smoke	0.30	Porstendorfer 1987
smoke	0.50	Porstendorfer 1987

light of the available information, F = 0.5 will continue to be employed as a nominal point estimate; an uncertainty range of 0.35 to 0.55 will be adopted.

c. Estimate of Average Occupancy Factor (Ω)

The occupancy factor, which is the percent of time spent in the home, naturally varies with lifestyle. For example, a study sponsored by the Electric Power Research Institute and conducted by GEOMET found values of 61.6%, 91.5%, and 86.2% for fully employed persons, housewives, and elderly individuals, respectively (GEOMET 1981). Exhibit 2-8 summarizes estimates of occupancy from the literature. For the most part, these estimates reflect analyses of detailed reports by individuals as to their activities over a fixed time period. However, it should be cautioned that several of the U.S. estimates are based on identical or overlapping data sets.

EXHIBIT 2-8
PERCENTAGE OF TIME SPENT AT HOME

Reference	Date	Country	Avg. Occupancy (%)
Oakley	1972	US	79.5
Moeller & Underhill	1976	US	74.4
GEOMET	1981	US	75.3
New Jersey	1989	US	71
U.S. EPA/OHEA	1989	US	64-73
U.S. EPA/ORP	1991b	US	61.5
Brown	1983	UK	75
Francis	1987	UK	77
Roy & Courtay	1990	France	76.7

The estimates for Ω vary from about 60% to 80%. The lowest value is the National Residential Radon Survey finding of 61.5%; this estimate is highly questionable, however, given the lack of detail obtained in the data acquisition and the concentration on summertime activity patterns. Based on the data in Exhibit 2-8, EPA will continue to employ an occupancy factor of 75%, as a nominal point estimate for calculating exposure and risk, with an uncertainty range of 65% to 80%.

d. Overall Uncertainty in the Estimate of Exposure

The estimated annual radon progeny exposure is proportional to the radon concentration (C), the equilibrium factor (F), and the occupancy factor (Ω). As discussed above, the average values for these parameters and the associated uncertainty ranges (in parentheses) are estimated to be: C =

1.25 pCi/L (1.1-1.4 pCi/L), F = 0.5 (0.35-0.55), and Ω = 0.75 (0.65-0.8). The geometric means of the uncertainty ranges are C_G = 1.24, F_G = 0.439, and Ω_G = 0.721. From Equation 8 it follows that the geometric mean of the uncertainty range for exposure is:

$$G_E = (0.242 \text{ WLM/y})\ (1.24/1.25)(0.439/0.5)(0.721/0.75) \tag{9}$$

Hence, G_E = 0.203 WLM/y. Treating the uncertainty distribution for each factor above as lognormal and the respective ranges as 90% confidence intervals, the corresponding 90% confidence interval for the uncertainty in exposure will be (G×L, G×U), where

$$\ln^2 U/L = \ln^2(1.4/1.1) + \ln^2(0.55/0.35) + \ln^2(0.8/0.65) \tag{10}$$

Solving this equation, U/L = 1.76. By symmetry of the lognormal distribution, U = 1/L = 1.32. Thus, the annual exposure is estimated to be between 0.203/1.32 and 0.203 × 1.32 WLM/y; i.e., from 0.154 to 0.268 WLM/y.

Numerical Estimates of Lung Cancers Induced by Residential Radon Exposures

Based on an average annual exposure of 0.242 WLM, the estimated number of radon-induced lung cancer deaths (lcd), per year, according to the modified BEIR IV model is:

$$N = (224 \text{ lcd per } 10^6 \text{ person-WLM})(0.242 \text{ WLM})(250\times10^6 \text{ persons})$$

$$\approx 13{,}600$$

An uncertainty range for N can be estimated from the calculated uncertainty distributions for the risk factor and for the annual exposure. The former was approximated by a lognormal distribution with geometric mean G_R = 283 lcd per 10^6 person-WLM and upper and lower 90% confidence bounds equal to $2.02\times G_R$ and $G_R/2.02$ (140 and 570). Similarly, for the latter, G_E was 0.203 WLM/y and the corresponding uncertainty bounds were $1.32\times G_E$ and $G_E/1.32$ (0.154 and 0.268 WLM/y). The upper and lower uncertainty bounds on N are then given by $N_L = 250\times10^6 (G_R\times G_E\times L)$ and $N_U = 250\times10^6 (G_R\times G_E\times U)$, where:

$$\ln^2 U = \ln^2 1/L = \ln^2 2.02 + \ln^2 1.32, \text{ or} \tag{11}$$

$$U = 1/L = 2.13. \tag{12}$$

The uncertainty distribution for N is then lognormal with geometric mean $G_N = 250\times10^6$, $(G_R\times G_E)$ = 14,400 lcd/y, and "90% confidence limits," $G_N/2.13$ and $2.13\times G_N$. Thus, it is projected that for the general population, 6,740 to 30,600 lcd/y are attributable to residential radon exposures.

EPA has continued to update its estimates of radon risk as exposure estimates have been refined and as new information has become available on the radon risk projection models and comparative dosimetry in mines and homes. Exhibit 2-9 shows the evolution of radon risk estimates over time, including the current estimate of approximately 7,000 to 30,000 lung cancer deaths per year due to residential radon exposure.

EXHIBIT 2-9 EPA'S RADON RISK ESTIMATES		
Model/Approach	Date of Estimate	Range of Estimated Annual Lung Cancer Deaths
EPA Model	1986	5,000 - 20,000
Average of ICRP 50 and BEIR IV	1988	8,000 - 43,000
BEIR IV Model (as adjusted by EPA)	1992	7,000 - 30,000

Section VI: The Effect of Smoking Status on Radon Risk

The revised *Citizen's Guide* includes charts on the risk of radon exposure for smokers and never smokers. The charts present risks in terms of lung cancer deaths/1,000 persons and also in comparison to the risk of death from different types of accidents. This section presents the approach used to estimate the lifetime lung cancer risk/person from radon for smokers and never smokers.

EPA estimates of annual lung cancer deaths due to radon presented earlier in this chapter are calculated for the general population, consisting of smokers and non-smokers. Since the risk of lung cancer from radon exposure appears to be enhanced by cigarette smoking, it is of interest to estimate the variation in radon risk by smoking category (current, former, and never smoker).

The assumption underlying the estimation of radon risk by smoking category is that radon risk varies in proportion to smoking risk. Therefore, for example, if lung cancer risk is more than twice as great in a current smoker compared to a person in the general population, then the radon risk for a current smoker also would be more than twice as great as that of a person in the general population. This assumption is consistent with the BEIR IV Committee conclusion that radon and smoking act multiplicatively in causing lung cancer (NAS 1988). For further discussion of the relationship between radon and smoking in causing lung cancer, see Section V of this chapter.

1. Smoking Risk Data

The data source for the prevalence and relative risks of smoking was the Surgeon General's Report, *Reducing the Health Consequences of Smoking* (DHHS 1989), which included the Cigarette Smoking Supplement to the 1985 National Health Interview Survey (NHIS 1985), and the American Cancer Society's Cancer Prevention Study II (American Cancer Society 1988). The National Health Interview Survey provided information on the prevalence of smoking in the U.S., and the American Cancer Society's CPS II study provided the relative risks for current and former smokers (see Exhibit 2-10).

To date, only the female data for CPS-II have been analyzed in detail; hence, estimates of risk for specific smoking levels (packs per day) or time since cessation cannot be made for males or the general population. For this reason, this analysis considers only the broad categories of current smokers and former smokers, with no further breakdown of smoking by subgroup.

It should be cautioned that the relative risks by smoking category derived in the CPS-II study may not accurately reflect the effect of smoking on lung cancer risk in the U.S. population. Subjects in CPS-II are more representative of middle-class white Americans than of the U.S. population as a whole. As a result, exposures to occupational and environmental carcinogens in the CPS-II group are likely to be lower than average. So, for example, non-smokers in the CPS-II group may have a lower incidence of lung cancer than non-smokers in general. In the case of current or former smokers, the study group may differ from the general population with regard to their smoking patterns, as well as other relevant factors.

2. Average Relative Risk

Exhibit 2-10 summarizes data from the Surgeon General's Report on smoking prevalence (proportion of the population in each smoking category) and relative risks of lung cancer by sex and by smoking category (current, former, and never smokers). These data were used to calculate a separate average relative risk of lung cancer for all males and for all females (see "All" under each gender in Exhibit 2-10). The relative risk of smoking is usually calculated as the ratio of the lung cancer mortality rate of a smoker group to that of a never smoker group. The average relative risk calculated here differs in that it is given by the ratio of the lung cancer mortality rate in the total

EXHIBIT 2-10
PREVALENCE AND LUNG CANCER RISKS OF CIGARETTE SMOKING

Category	Prevalence[a]	Relative Risk[b]	Lung Cancer Deaths/10^5 persons[c]
Males			
Current smoker	0.327	22.36	14,118
Former smoker	0.291	9.36	5,909
Never smoked regularly	0.382	1.0	631.4
All (average)[d]	1.0	10.42[c]	6,579
Females			
Current smoker	0.275	11.92	5,593
Former smoker	0.171	4.69	2,200
Never smoked regularly	0.554	1.0	469.2
All (average)[d]	1.0	4.64[c]	2,177

[a] Proportion of population in each smoking category; data from Surgeon General's Report (DHHS 1989).
[b] Defined as the ratio of the age-adjusted lung cancer mortality rate in the current and former smoker category to (divided by) the age-adjusted lung cancer mortality rate in the never smoker category; data from Surgeon General's Report (DHHS 1989).
[c] Calculated by EPA, as described in text.
[d] Includes current, former, and never smokers.

male or female population (including never smokers) to the lung cancer mortality rate in the never smokers for each respective gender. This approach to the calculation of relative risk is based on the methodology employed in the National Institutes of Health report on the development of radioepidemiological tables (NIH 1985).

Thus the average relative risk for all males is:

$$\overline{RR} = \sum_i p_i \, RR_i \tag{13}$$

where p_i and RR_i are the prevalences and relative risks for all smoking categories including those persons who never smoked regularly. Thus, the average relative risk for all males is

$$\overline{RR} = 0.327 \times 22.36 + 0.291 \times 9.36 + 0.382 \times 1.0 = 10.42$$

relative to males who never smoked regularly. Similarly, for all females, the average relative risk is

$$\overline{RR} = 0.275 \times 11.94 + 0.171 \times 4.69 + 0.554 \times 1.0 = 4.64$$

relative to females who never smoked regularly.

3. Lung Cancer Mortality Calculations

The average relative risks calculated in Section 2 were used in conjunction with 1980 vital statistics and decennial life tables (NCHS 1983, NCHS 1985) to calculate lung cancer deaths in male and female birth cohorts (all smoking categories combined) of 100,000. The number of lung cancer deaths yielded by these calculations was 6,579 and 2,177 per 100,000 males and per 100,000 females, respectively (see Exhibit 2-10).

The number of lung cancer deaths in cohorts of 100,000 of each smoking category of males and of females was then calculated. The lung cancer death rate for each category was assumed to be proportional to the relative risk for that smoking category of males or females. For example, the lung cancer death rate in male current smokers (M,CS) was given by:

$$\begin{aligned} \text{lcd(M,CS)} &= (22.36/10.42)\ (6579\ \text{lcd}/100{,}000) \\ &= 14{,}118\ \text{lcd}/100{,}000. \end{aligned}$$

The estimated lung cancer deaths/100,000 for all smoking categories and both sexes are listed in Exhibit 2-10.

For the general population (defined as males and females combined), the number of lung cancer deaths was subsequently estimated by adjusting the number of lung cancer deaths calculated for males and for females according to the sex distribution of the birth cohort (1.051 male: 1 female). Thus, the number of lung cancer deaths for the general population would be (1.051x6579 + 2177)/2.051 = 4433 lung cancer deaths (see Exhibit 2-11). To calculate the number of lung cancer deaths in each smoking category of the general population, the male and female lung cancer rates were combined. As an example, consider the case of current smokers. The fraction of the current smokers birth cohort that is male was estimated to be:

$$f(M,CS) = \frac{1.051\ p(M,CS)}{1.051\ p(M,CS) + p(F,CS)} \qquad (14)$$

where p(M,CS) and p(F,CS) are the prevalences of current smokers among males and females, respectively. From Exhibit 2-10, p(M,CS) = 0.327 and p(F,CS) = 0.275. Solving, f(CS,M) = 0.5555. The fraction of current smokers that is female is then given by:

$$f(CS,F) = 1\text{-}f(CS,M) = 0.4445 \qquad (15)$$

In a birth cohort of 100,000 (male and female) current smokers, therefore, the number of lung cancer deaths would be:

$$lcd(CS) = (0.5555)(14{,}118)+(0.4445)(5593) = 10{,}329$$

Lung cancer deaths in birth cohorts of 100,000 (male and female) former smokers and 100,000 (male and female) never smokers were calculated following the same approach, and are presented in Exhibit 2-11.

EXHIBIT 2-11
LUNG CANCER RISK OF CIGARETTE SMOKING IN THE GENERAL POPULATION[a]

Category	Lung Cancer Deaths/10^5persons
Current Smoker	10,329
Former Smoker	4,579
Never Smoked Regularly	537.4
All (Average)	4,433

[a] Combined male and female data from Exhibit 2-10. Sex ratio at birth, M:F = 1.051:1.

The lung cancer death rates in each smoking category in the general population were compared to the average lung cancer death rate for the general population to obtain the radon risk multipliers presented in Exhibit 2-12. For example, the number of lung cancer deaths/100,000 current smokers in the general population is 10,329. This is a factor of 2.33 times the 4,433 lung cancer deaths expected in the general population, averaging over all smoking categories. It follows from the presumed multiplicative interaction between radon and smoking that the radon risk among current smokers also would be about 2.33 times the radon risk for the general population. Risk multipliers were derived for former and never smokers as well, using this approach. As shown in Exhibit 2-12, the lung cancer risk (and thus the radon risk) for never and former smokers in the general population are estimated, respectively, to be 0.121 and 1.03 times that of the general population.

EXHIBIT 2-12
RISK MULTIPLIERS BY SMOKING CATEGORY
IN THE GENERAL POPULATION[a/]

Category	Risk Multiplier
Current Smoker	2.33
Former Smoker	1.03
Never Smoked Regularly	0.121
All (Average)	1.00

[a/] Calculated in text.

4. Lifetime Radon Risk by Smoking Category

The above risk multipliers were used in conjunction with a standard lifetable analysis, 1980 decennial vital statistics and the EPA-adjusted BEIR IV relative risk model ($K = 0.7$) to estimate the lung cancer risk for never smokers and for current smokers at several radon levels. As before, in using the BEIR IV model, the lung cancer baseline risk was adjusted for an annual background radon exposure of 0.242 WLM.

For a more rigorous calculation of the excess risk due to radon, the baseline lung cancer risk for current smokers and for never smokers would be multiplied by the risk multipliers 2.33 and 0.121, respectively. This would mean constructing new lifetables for both groups. However, there are a number of difficulties with this approach. For example, age-specific death rates are generally not available for smokers and never smokers separately. A simpler approach is to multiply the excess lung cancer risks (not the baseline risks) due to radon for never smokers and for current smokers by their respective risk multipliers. For example, in the case of current smokers, the excess risk $R(a)$ is estimated as:

$$R(a) = 2.33 \times [r(a) - r_0(a)] \qquad (16)$$

where $r(a)$ is the age dependent total lung cancer risk and $r_0(a)$ is the age dependent baseline lung cancer risk. Simplifying the calculation gives risk estimates that are within 10% of the values calculated using the same approach described in the BEIR IV report (NAS 1988), without requiring changes to the lifetables.

Exhibit 2-13 shows the risks for never smokers and current smokers along with the risks for the general population, for selected exposure levels. Levels are expressed in both pCi/L and WLM/y. The relationship between the two is as follows:

$$\text{WLM/y} = [F \times 0.01\ \text{WL/(pCi/L)}] \times [\Omega \times 51.6\ \text{WLM/WL-y}]\ \text{pCi/L} \qquad (17)$$

where:

F, the equilibrium factor, is assumed to be 0.5, and

Ω, the fraction of time spent indoors, is assumed to be 0.75.

For further discussion of the average equilibrium factor and average occupancy factor, see Section V.

EXHIBIT 2-13
LIFETIME LUNG CANCER RISK FOR NEVER SMOKERS, CURRENT SMOKERS, AND THE GENERAL POPULATION

Radon Level		Lifetime Lung Cancer Risk (per person)		
pCi/L	WLM/y	Never Smokers	Current Smokers	General Population
20	3.87	7.72×10^{-3}	1.35×10^{-1}	6.15×10^{-2}
10	1.94	3.87×10^{-3}	7.11×10^{-2}	3.14×10^{-2}
8	1.55	3.10×10^{-3}	5.74×10^{-2}	2.52×10^{-2}
4	0.774	1.55×10^{-3}	2.93×10^{-2}	1.27×10^{-2}
2	0.387	7.76×10^{-4}	1.48×10^{-2}	6.39×10^{-3}
1.25	0.242	4.85×10^{-4}	9.29×10^{-3}	4.00×10^{-3}
0.40	0.077	1.55×10^{-4}	2.98×10^{-3}	1.28×10^{-3}

5. Discussion

The approach used here for calculating the risks for various smoking categories is approximate. It does not account for life-shortening due to smoking, which reduces the relative contribution of smoking-attributable lung cancer deaths. An analysis of the effect of smoking presented in BEIR IV only partly corrected for life-shortening because it neglected all smoking-related causes of death other than lung cancer. EPA calculations show that use of the BEIR IV methodology would produce a difference of less than 10% in the risk multipliers presented here. In light of the larger potential sources of error associated with the calculation, these differences would appear to be insignificant. The chief sources of error and uncertainty associated with the analysis of the effects of smoking on radon risk are the uncertainty over the actual form of the interaction (e.g., multiplicative or submultiplicative); the variations in the relative risk of smoking by age and gender;

the changes in age-specific relative risks for smokers as smoking habits and types of cigarettes change over time; and the influence of environmental and passive cigarette smoke on radon risk.

Section VII: Conclusions

For most pollutants, the assessment of human risk is based on data from animals exposed to very high levels of a given substance. The large and obvious uncertainty associated with this extrapolation is usually compounded by the fact that the substance may be administered to laboratory animals in a way that is different from human exposure (e.g., oral versus inhalation). In addition, the animals are often exposed to levels of a substance several orders of magnitude greater than typical human exposures.

Radon is a known human carcinogen. Estimates of lung cancer risk for the general population from inhalation of radon and radon progeny are derived using human data on the occurrence of lung cancer in underground miners due to inhalation exposures. Furthermore, the average lifetime cumulative radon exposures to the general population are only slightly below those for which increased risk can be demonstrated in underground miners. Hence, low-exposure extrapolation is less of an issue for radon risk assessment than it is for the assessment of risk for most other environmental carcinogens.

There is extensive evidence supporting the classification of radon as a known human carcinogen. There is some uncertainty associated with the projection of lung cancer risk from occupational radon exposures to the general population for residential exposures. Although the epidemiologic data on increased risk from indoor exposures are limited, some of the studies suggest that the relative risk coefficients for the general population and for underground miners are comparable.

Uncertainty is always present to some extent in risk assessment for environmental pollutants. However, as more data become available for radon, the uncertainties associated with estimation of lung cancer risk may diminish. Follow-up of the underground miners will continue to clarify the relationship of radon and smoking and the nature of the exposure-response relationship for radon exposure and lung cancer risk over a lifetime. The results of epidemiologic studies of indoor radon exposure may help to further define the relationship at low exposure levels. Advances in dosimetric modeling and the findings of the NAS' report on comparative radon dosimetry (NAS 1991) have improved the extrapolation of risks from miners to the public. As new information becomes available, EPA will continue to update its estimates of lung cancer risk from residential radon exposure.

CHAPTER 3

ANALYSIS OF EPA RADON TESTING OPTIONS

EPA recommends that individuals living in the following types of housing test their homes for radon: single family homes, apartment units within multi-unit structures (if they are below the third floor), mobile homes with permanent foundations, and units in group quarters below the third floor (e.g., college dormitories). Among other changes, the proposed revision of the *Citizen's Guide* recommends a new procedure for testing home radon levels. EPA narrowed the options down to six and chose a final testing procedure from among those six approaches.[1] This chapter describes the six options and explains the process the Agency used to analyze all of them before recommending one for the public to use.

THE AIMS OF THIS CHAPTER

(1) To describe the options for radon testing procedures EPA considered recommending to the public.

(2) To explain how EPA analyzed the effectiveness of these options in correctly categorizing homes above or below the threshold requiring radon mitigation.

This chapter is organized into two major parts. The first provides background information, reviews the major considerations important to developing testing procedures, and summarizes the results of an analysis of the effectiveness of the testing options (Sections I, II, and III). The second part of the chapter presents the details of the analysis and closes with general conclusions drawn from the analytic results (Sections IV, V, and VI). This second part describes the analytic process, but does not replicate the analysis. Therefore, EPA has cited rather than included technical background information. Finally, Appendix C supports this chapter by providing basic background information on commonly used measurement devices.

Section I: Background

The original *Citizen's Guide* recommended that individuals perform (1) a short-term (screening) measurement in the lowest livable level of the home under "closed-house" conditions during the winter months, and (2) confirmatory long-term tests in the lived-in levels of the home if the result of the short-term test was above 4 picocuries per liter (pCi/L). The recommended duration of the confirmatory tests depended on the short-term test results and varied from less than one week to a full year. The short-term test was designed to obtain the most reproducible estimate of the highest (worst-case) radon concentration likely to be found in that particular house at any time. The long-term measurement was intended to be an estimator of the annual average concentration in lived-in levels of the home, being made over an entire year under normal ventilation conditions. An exception to this process was when the short-term result was so elevated as to necessitate a more rapid confirmation. Descriptions of short-term and long-term tests are given below.

[1]EPA originally considered more than six options. However, for reasons discussed in Section II (Appropriate Testing Location), EPA focused this analysis on the six options that recommend testing on the lowest lived-in level.

Short-term measurements, which last from 2 to 90 days, aim to give a quick assessment of the level of radon in a home. Short-term screening measurements, which EPA recommended be taken in the winter months in the "lowest livable" level of homes (the bottom-most inhabitable space) under "closed-house" conditions, tend to produce conservative (i.e., high) estimates of annual average radon levels. Short-term tests also can be conducted in the lowest lived-in level of homes (a room that is used regularly) to produce estimates that more closely represent the annual average radon concentration.

Long-term measurements are usually taken over the course of an entire year under normal ventilation conditions. Because long-term measurements take into account seasonal fluctuations in radon levels, and because they are performed under the ventilation conditions normally encountered by homeowners, they tend to be better indicators of the actual radon concentrations to which individuals in a home are exposed.

While long-term measurements tend to be better indicators of the average radon concentration in a house over the course of a year, short-term measurements are simpler to make and provide faster results. In cases where radon concentrations are extremely high, fast results are desirable in order for mitigation to begin quickly.

An individual's risk of getting lung cancer from radon is directly proportional to his or her integrated exposure, and the annual average concentration in lived-in levels of the home is the most readily measurable quantity that approximates that exposure. However, it is not necessary to know the actual risk in order to act to reduce it; a homeowner only needs to know that the radon concentration is above some action level. Since the public seemed willing to take action based only on short-term tests (see Chapter 6), EPA needed to adopt a recommended testing protocol for the revised Guide that would allow homeowners to make the best possible mitigation decision based on short-term test results.

In revising the *Citizen's Guide*, EPA has focused on how well options meet the basic purpose of testing -- to enable homeowners to make the right decision on whether to mitigate radon levels. When analyzing those options utilizing short-term measurements, the primary concern was not on how well short-term measurements could estimate long-term averages. Rather, the central issue was how well a testing option provides a reasonably accurate basis for determining whether there is a need to reduce radon levels in a home. Errors could result when (1) a test indicated that mitigation should occur when actual radon levels were below the action level (false positive), or (2) a test indicated that mitigation should not occur when actual radon levels were above the action level (false negative). Combining the percentages of both types of errors provides an estimate of the rate to which each option might misclassify homes for mitigation. As discussed in the next section, the misclassification rate for various testing schemes was one of five major factors that was important to EPA in evaluating testing options.

Section II: Major Considerations in Developing Testing Procedures

EPA's goal is to recommend a radon testing procedure in homes that will lead the public to decisions about radon reduction that are timely, simple, cost-effective, and protective of human health. Before developing possible testing procedures, EPA identified five major factors that were important to consider in developing and evaluating options. These considerations are listed in summary form in Exhibit 3-1. The first factor is the action level used to determine whether a home

EXHIBIT 3-1
CONSIDERATIONS IN DEVELOPING RADON TESTING OPTIONS

- Appropriate Action Level
- Appropriate Testing Location
- Appropriate Ventilation Conditions
- Acceptability to Potential Users
- Effectiveness of Short-Term and Long-Term Measurements

requires mitigation. The second concern is the location in the home where testing occurs. The third factor is the ventilation condition (i.e., "open-house condition" versus "closed-house condition") under which tests are performed. The fourth concern is the acceptability of the procedure to potential users, and the final factor is the effectiveness of relying on short-term measurements. Each of these factors is explained in more detail in the text that follows.

Appropriate Action Level — EPA is recommending an action level of 4 pCi/L for several reasons. First, the Office of Research and Development's (ORD's) research on mitigation effectiveness and the Office of Radiation Programs' mitigator survey suggest that elevated levels of radon can be reduced to 4 pCi/L more than 95 percent of the time. Results from the mitigator survey indicate that 2 pCi/L can be achieved about 70 percent of the time, while the ORD research suggests this estimate may be even higher (U.S. EPA/ORD 1989; U.S. EPA/Radon Division 1990a). Reducing the action level to 2 pCi/L, therefore, could result in perhaps as many as 30 percent of homes with elevated levels being unable to achieve the action level. Second, lower action levels introduce more uncertainty in the measurement results. Measurement device error increases to approximately 50 percent at 2 pCi/L. This device error in conjunction with the larger fraction of homes (of total homes testing) that have radon levels around 2 pCi/L would result in a threefold increase in false negatives and a twofold increase in false positives over those expected at a 4 pCi/L action level.

Appropriate Testing Location — In the 1986 *Citizen's Guide*, EPA recommended that, for screening purposes, a short-term measurement be taken on the "lowest livable" level, i.e., the lowest area of a home that is used or has the potential to be used as a living space during the winter months. This definition would include an unfinished basement that is in a condition such that it could be converted to a bedroom, playroom, den, etc. For a confirmatory test (if the short-term test result was above 4 pCi/L), the 1986 *Guide* recommended a follow-up test in the lived-in levels of the home. Lived-in level is defined as a level of a house that is used regularly, such as a living room, playroom, den, or bedroom, but not a kitchen or bathroom, where high humidity conditions or the operation of an exhaust fan could affect the validity of the test. EPA considered both "lowest livable" and "lowest lived-in" testing locations in formulating the testing protocol in the revised *Guide*.

The choice between "lowest livable" and "lowest lived-in" level affects where homes with basements will place radon detectors. Roughly 50 percent of homes nationwide have basements; only half of these homes, or almost 25 percent of the national total, have basements that are used as a lived-in level (U.S. EPA/Office of Radiation Programs 1991a). Accordingly, a recommendation that devices be placed in the "lowest livable" level would result in 50 percent of the devices being placed in the basement. A recommendation that devices be placed in the "lowest lived-in" level would result in only 25 percent of devices being placed in the basement, and the remaining 75 percent being placed on the first floor. A difference in the testing location therefore would affect 50% of homes with basements that are not presently lived-in; this is 25% of all U.S. homes.

Keeping the short-term test in the lowest livable area as compared to moving it to the lowest lived-in area would reduce false negative results by a factor of 1.5; however, the number of false positive results would increase by a factor of 2. Measurements taken in the lowest lived-in area strike a more equitable balance between false positives (public money spent with more limited benefits in many cases) and false negatives (public health protection) than do livable area measurements. Additionally, recent research (Harley et al. 1991) suggests that basement short-term measurements overstate personal exposure by a factor of 3 to 5, while first floor short-term measurements are, on average, only 30 percent greater than occupant exposure. Since the goal of radon measurement on which a mitigation decision relies is to assess occupant exposure (U.S. EPA 1991a), lived-in level measurements are better predictors than livable level measurements. Based on this finding and the recommendation of EPA's Science Advisory Board (SAB) (U.S. EPA 1992b), short- or long-term measurements that are to be used for the purpose of making mitigation decisions should be made on the lowest lived-in level since it more closely approximates that concentration to which inhabitants are exposed. Therefore, only lived-in level testing is considered in the various testing options discussed below.

Appropriate Ventilation Conditions — EPA examined whether tests should be made under "open-house" or "closed-house" conditions. In open-house conditions, tests are made with windows and other ventilating passageways either closed or open as they would be when a test is not being conducted. In closed-house conditions, all windows and other ventilated passageways are closed, to the extent possible. Overall, EPA stresses the value of closed-house conditions in the final revised *Guide*. Specifically, the final *Guide* recommends keeping windows and outside doors closed "as much as possible" during short-term testing. For short-term tests lasting just 2 or 3 days, the final *Guide* recommends closing windows and outside doors at least 12 hours before beginning the test.

Acceptability to Potential Users — EPA realized that, in real terms, the testing procedure recommended in the original *Citizen's Guide* was technically sound, but was not viable because many individuals dropped out before obtaining long-term measurements of the radon level in their homes. Current Radon Program experience indicates that long-term tests are a deterrent to action and that few people take long-term follow-up tests (see Chapter 6). In fact, the majority of people who mitigate their homes are currently doing so based on a single short-term measurement.

Effectiveness of Short-Term and Long-Term Tests — Long-term tests are more indicative of annual exposure than are short-term tests. Although a few of the testing procedures incorporate the use of long-term tests, experience indicates that people are unwilling to take such tests. (Please refer to Chapter 6 for more details on the public's unwillingness to use long-term tests.) Because short-term tests are completed in a short period of time, they are much more appealing to the public. A procedure that does not incorporate long-term testing may be more effective in actual risk reduction. Therefore, EPA assessed the feasibility of testing procedures utilizing short-term tests. Before recommending a testing procedure not based on long-term testing, EPA sought to examine the implications of relying on short-term tests as indicators of whether or not mitigation is necessary.

From knowledge of the relationship between short-term measurements and annual averages gained from EPA/State Residential Radon Surveys (SRRS), the Agency believed that although short-term measurements are imperfect indicators of annual averages, they could potentially serve effectively in the intended use EPA was considering. EPA reasoned that people measuring radon levels in their homes do not require an accurate measurement of the annual average; instead, homeowners require only a reasonably accurate basis for determining whether they need to mitigate. Testing options need to have low misclassification rates for mitigation decisions to be acceptable. EPA recognized that the distribution of radon levels in homes that is believed to exist would work in the Agency's favor in using short-term tests. Exhibit 3-2 shows the distribution of annual average radon concentrations suggested by EPA's National Residential Radon Survey (NRRS) for homes that should be tested according to EPA's recommendations. Over 82 percent of the homes have radon levels below 2 pCi/L. Another 7 percent of the homes have levels between 2 and 3 pCi/L. (These estimates are based on lived-in level averages.) Therefore, errors in the use of short-term testing for nearly 90 percent of the homes would have to be very substantial for these homes to be misclassified as above the action level. The largest chances for problems exist in the range of 3 to 5 pCi/L, where there are roughly 6 percent of all houses. The skewed distribution of radon levels towards the lower end of the range could serve as a strong moderating force to any potential misclassification errors.

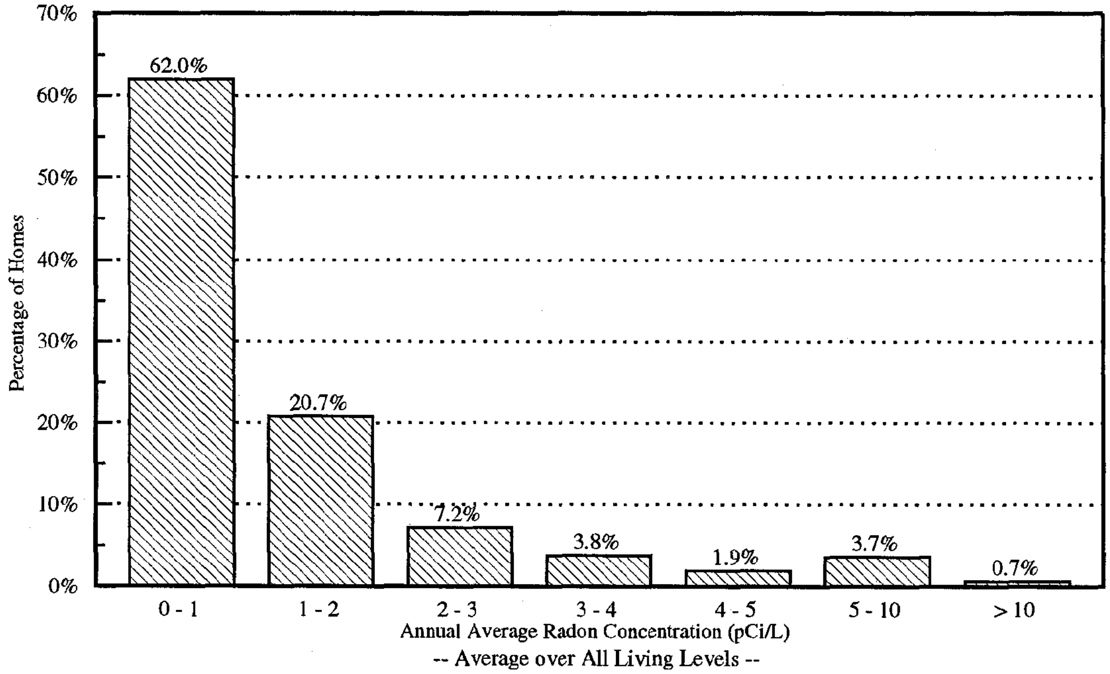
EPA was able to assess how short-term measurements could be used to make radon mitigation decisions, as it had carefully tested large numbers of homes and compiled data to enable such an evaluation. Beginning in 1987, EPA and States had cooperatively conducted 40 state-wide surveys of radon levels in homes. For these surveys, which were designed to eliminate bias using standard statistical techniques, EPA collected short-term radon measurements from nearly 60,000 homes and paired short-term/long-term measurements from over 1,000 homes. EPA used testing protocols and maintained a strong quality assurance and control program to ensure that the radon testing data collected could be relied on in EPA evaluations of the significance of the radon problem and in determining recommendations for public action.

As explained in more detail later in this chapter, EPA used both the SRRS and NRRS results in a variety of ways in the analysis of radon testing options. For example, the Agency used the SRRS results to analyze the relationship between short- and long-term measurements. The Agency used the NRRS results to determine the distribution of annual average radon concentrations in homes across the nation.

EPA initially considered the implications of a single, short-term test that people would conduct in the lowest lived-in level of their homes. This simple approach did not offer enough confidence that homes would be correctly classified in relation to the action level. Therefore, EPA concentrated on analyzing options that relied on multiple tests. One of the options was very similar to the short-term/long-term testing approach in the original *Citizen's Guide*, except the test would occur in the lowest lived-in level (as opposed to the lowest livable level). This option is a reliable approach that can be compared to options using short-term confirmatory tests. EPA evaluated five other options that relied on multiple tests in a shorter time interval than the first option. All six options were analyzed to estimate their rate of misclassification of homes for mitigation.

EXHIBIT 3-2

DISTRIBUTION OF RADON LEVELS IN U.S. HOMES FOR WHICH EPA RECOMMENDS RADON TESTING



Source: U.S. EPA/Office of Radiation Programs 1991a.

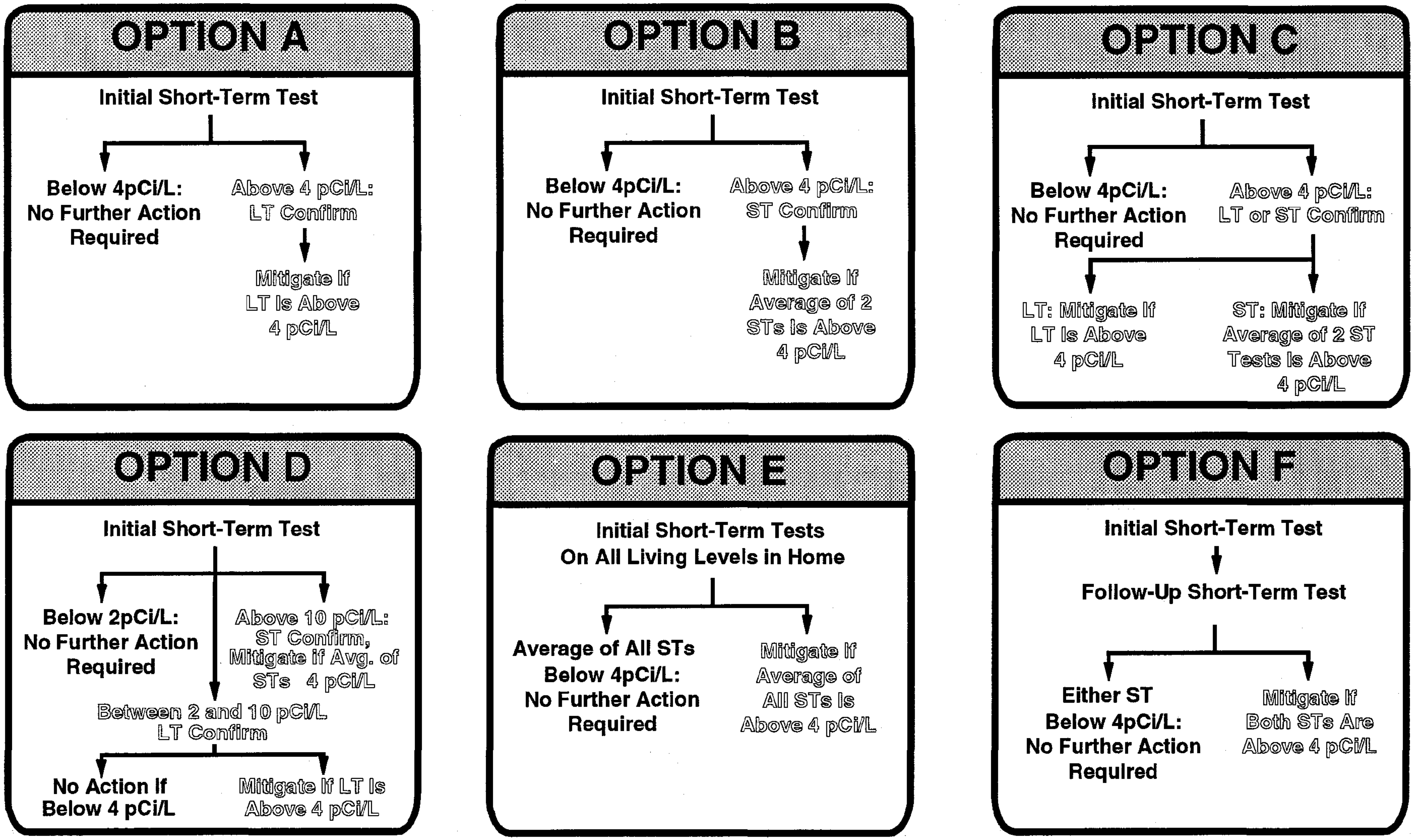
Section III: Summary of the Misclassification Analysis Results for EPA's Testing Options

This section describes the six testing options that the Agency considered in developing a testing protocol for the revised *Citizen's Guide*. It then describes the criteria used to judge the effectiveness of each option, and presents the results of the Agency's analysis in summary form. The options considered are diagrammed in Exhibit 3-3. For this document, EPA labeled the six options in the order that makes their presentation most comprehensible. The Agency did not necessarily create or evaluate options in the order in which they are described in this chapter.

When describing testing alternatives, the Agency made a distinction between **action level** and **trigger level.** The action level was the level above which EPA recommended that a home be mitigated. It also could be the same point where additional testing should occur. In contrast, the trigger level was the level above which the Agency recommended in Option D that an individual conduct further long-term or short-term testing to determine if a home was above the action level. The action level was fixed at 4 pCi/L. The trigger levels in Option D were fixed at 2 pCi/L (additional long-term testing required) and 10 pCi/L (additional short-term testing required).

EXHIBIT 3-3

ILLUSTRATION OF TESTING OPTIONS CONSIDERED



ST = Short-Term Test   LT = Long-Term Test

- Option A — Individuals would make an initial short-term measurement in the lowest lived-in level. If the measurement exceeded the action level set by EPA, the individual would be advised to make a long-term (i.e., year-long) measurement. If this measurement exceeded the action level set by EPA, the Agency would recommend mitigation. [Note: The original *Citizen's Guide* recommended a testing procedure similar to Option A, except that short-term tests were conducted on the lowest livable level, and the recommended duration of the confirmatory test was not necessarily one full year.]

- Option B — Individuals would make an initial short-term measurement in the lowest lived-in level. If the measurement exceeded the action level, the individual would be advised to make a confirmatory short-term measurement. If the average of the two short-term measurements exceeded the action level set by EPA, the Agency would recommend mitigation. [Note: EPA proposed this testing procedure in the revised draft of the *Citizen's Guide*, published in 1990.]

- Option C — Individuals would make an initial short-term measurement in the lowest lived-in level. If the result exceeded the action level, the individual would make either a confirmatory long-term or short-term measurement. This essentially gives individuals the choice of Options A and B. If a confirmatory long-term measurement were made, the results would be compared to the EPA action level, as in Option A. If a confirmatory short-term measurement were made, the average of the two short-term tests would be compared to the EPA action level, as in Option B. If either method resulted in a value which exceeded the action level, the Agency would recommend mitigation.

- Option D — Individuals would make an initial short-term measurement in the lowest lived-in level of their home. If the measurement exceeded a trigger level of 10 pCi/L, the individual would be advised to make a short-term test to confirm that the home required mitigation. If the measurement was below 10 pCi/L but above a second trigger level of 2 pCi/L, the individual would be asked to make a confirmatory long-term measurement. If the average of the two short-term tests (for homes above 10 pCi/L) or the result of the long-term measurement exceeded the action level, the Agency would recommend mitigation.

- Option E — Individuals would make simultaneous short-term measurements on all lived-in levels of their home, one measurement device per level. For single level homes, the Agency would recommend that individuals perform two tests in different rooms. If the average of all test results indicated that the radon level in the home exceeded the action level, the Agency would recommend mitigation.

- Option F — Like Option B, this option requires that individuals make an initial short-term measurement in the lowest lived-in level. If the measurement exceeded the action level, the individual would be advised to make a follow-up short-term measurement. If the confirmatory measurement also exceeded the action level, the Agency would recommend mitigation.

EPA used the following six quantitative measures to evaluate the simplified testing procedures. Each measure was expressed as a percentage of the total homes tested, except the fifth and sixth measures, which are explained further below.

Correct classification percentage — the percentage of homes tested for which the testing procedure came to the correct conclusion (i.e., that homes with high radon levels should be mitigated and all others left alone).

Error rate — homes for which the testing procedure came to the incorrect conclusion (i.e., either false negative or false positive). This is the misclassification rate.

False negatives — homes tested in which the procedure incorrectly concluded that mitigation was not required. False negatives can be presented as a percent of all homes or as a percent of actual positives (i.e., homes that actually have radon levels greater than 4 pCi/L regardless of test results).

False positives — homes tested in which the procedure incorrectly concluded that mitigation was required. False positives can be presented as a percent of all homes or as a percent of all positive tests.

False positives with quantifiable benefits — homes with false positive test results with actual annual radon levels between 2 and 4 pCi/L (i.e., homes that have low starting radon concentrations, but will still receive significant risk reduction through mitigation). False positives with quantifiable benefits are presented as a percent of all positive tests.

False positives with non-quantifiable benefits — homes with false positive test results with actual annual radon levels below 2 pCi/L. These homes also will also receive risk reduction through mitigation, though the risk reduction will be less than that in homes with actual annual radon levels between 2 and 4 pCi/L. Because the cost-effectiveness analysis presented in Chapter 5 assumes that mitigation only lowers homes to an average of 2 pCi/L, the actual benefits of fixing homes with initial levels below 2 pCi/L is left unquantified. False positives with these "non-quantifiable" benefits are presented as a percent of all positive tests.

Exhibit 3-4 schematically shows how misclassification rates depend on the annual average lived-in area radon concentration, the result of the testing procedure, and the action level. Among other items, the exhibit shows that the inaccuracies in testing procedures matter only when they produce results on the opposite side of the action level from the correct annual average (i.e., misclassifications). The exhibit also shows that the correct classification percentage and the error rate are complementary, and that the false negative percentage and the false positive percentage add up to the error rate. In the remainder of this chapter, EPA refers to the error rate as the misclassification rate.

To determine the effectiveness of testing procedure options, EPA performed statistical analyses of the misclassification rates associated with each option. EPA was interested in how frequently a given test procedure correctly indicated whether a home should be mitigated. Exhibit

EXHIBIT 3-4
THE ACCURACY OF MITIGATION DECISIONS BASED ON SHORT-TERM MEASUREMENTS

	Annual Average < Action Level	Annual Average ≥ Action Level
Short-Term Measurement < Action Level	CORRECT CLASSIFICATION Although the short-term measurement may vary from the annual average, both are below the action level. The short-term test correctly indicates that mitigation is unnecessary.	MISCLASSIFICATION *False Negative* — the short-term test incorrectly indicates that mitigation is unnecessary.
Short-Term Measurement ≥ Action Level	MISCLASSIFICATION *False Positive* — the short-term test incorrectly indicates that mitigation is necessary.	**CORRECT CLASSIFICATION** Although the short-term measurement may vary from the annual average, both are at or above the action level. The short-term test correctly indicates that mitigation is necessary.

Correct Classification Percentage (corresponds to total unshaded cells) = $\frac{\text{Number of times testing led to a correct conclusion}}{\text{total homes tested}}$

Error Rate (corresponds to total shaded cells) = $\frac{\text{Number of times testing led to an incorrect conclusion}}{\text{total homes tested}}$

Note: Often home mitigations will reduce radon levels to 2 pCi/L or lower. Consequently, mitigations undertaken due to false positive short-term testing results where the annual average radon concentration is above 2 pCi/L (but less than 4 pCi/L) may result in significant risk reduction.

3-5 summarizes the results of EPA's statistical analysis of the six testing procedures it considered. As recommended by the SAB (U.S. EPA 1992b), all results are based on the true radon levels being equal to the annual average radon levels in homes, across all lived-in levels, as determined by EPA's National Residential Radon Survey (U.S. EPA/Office of Radiation Programs 1991a). Results are based on an action level of 4 pCi/L (the level used in both the original and revised *Citizen's Guide*).

EXHIBIT 3-5
TESTING PROCEDURES EPA CONSIDERED AND THEIR MISCLASSIFICATION RATES
(Action Level = 4 pCi/L)

Test Accuracy	Option A	Option B	Option C	Option D	Option E	Option F
Correct Classification	98.1%	93.7%	94.1%	98.5%	94.7%	94.8%
Error Rate	1.9%	6.3%	5.9%	1.5%	5.3%	5.2%
False Negatives as a Percent of All Tests	1.9%	2.3%	2.3%	0.5%	1.5%	2.8%
False Negatives as a Percent of Actual Positives	30.2%	36.5%	36.5%	7.9%	23.8%	44.4%
False Positives as a Percent of All Tests	0.0%	4.0%	3.6%	1.0%	3.8%	2.4%
False Positives as a Percent of All Positive Tests	0.0%	49.9%	47.4%	15.2%	44.3%	40.4%
False Positives with Quantifiable Benefit	0.0%	28.9%	27.4%	8.2%	30.9%	26.5%
False Positives with Non-Quantifiable Benefit	0.0%	21.0%	20.0%	7.0%	13.4%	13.9%

The accuracy of the testing procedure options also can be expressed in terms of the annual lung cancer deaths that would be averted (i.e., the annual lives that would be saved) and the annual deaths from lung cancer that would not be prevented (i.e., the number of lung cancer deaths in homes that received a false negative test result). For the purposes of these calculations, EPA assumed that 100 percent of the population followed the testing procedures and mitigated their homes when results were above the action level (see Appendix E for a more detailed explanation). Exhibit 3-6 presents the annual lives saved due to mitigations (where risk reductions were quantifiable) under each of the testing procedure options. The exhibit also shows the lives not saved due to false negative testing results.

EXHIBIT 3-6
ANNUAL LIVES SAVED AND NOT SAVED
UNDER TESTING PROCEDURE OPTIONS[a]

Testing Option	Annual Lives Saved in Homes with True Positive Results	Annual Lives Saved in Homes with False Positive Results	Total Lives Saved Annually	Annual Lives Not Saved in Homes with False Negative Results
A	2,160	0	2,160	620
B	2,040	200	2,240	750
C	2,050	180	2,230	740
D	2,630	50	2,680	160
E	2,330	240	2,570	450
F	1,820	140	1,960	970

[a] Estimates assume 100% testing and mitigation.

Section IV: Approach for Analyzing Misclassification Rates of Each Testing Procedure Option

To help determine which testing procedure option to recommend to the public, EPA focused on how well testing options provided a reasonably accurate basis for determining whether there was a need to mitigate radon levels in a home. EPA wanted to estimate the misclassification rates associated with each option. Misclassifications result from the inaccuracies of test measurements in estimating the annual average with respect to the action level.

Because even Option A, which relies only on long-term tests for confirmatory purposes, relies on a short-term test for initial screening, all of the options EPA considered have misclassifications resulting from short-term testing. To analyze the misclassification rates that result from the short-term testing components of each of the options, the Agency used a statistical approach that relied to some extent on empirical data. The six steps involved in this statistical approach are summarized in Exhibit 3-7. These steps are explained in more detail in the text that follows according to the step numbering in the exhibit.[2]

[2] The development of EPA's statistical approach is further documented in two reports. The first report describes the Agency's empirical analysis of existing radon measurement data (Marcinowski 1990). The second report details EPA's development of a quantitative approach to predict short-term misclassification rates (Chmelynski 1992).

EXHIBIT 3-7
STEPS INVOLVED IN EPA'S
STATISTICAL ANALYSIS OF TESTING OPTIONS

(1) EPA developed a statistical model that explained a short-term test's inaccuracy in estimating the annual average.

(2) For each option, EPA determined what test procedure outcomes were possible, depending on the results of the initial and confirmatory tests.

(3) EPA considered the misclassifications resulting from all possible outcomes for a given option separately.

(4) To determine how many homes would not require a confirmatory test and how many false negatives there would be among homes not retesting, EPA modeled the outcome of the initial test by developing a joint probability distribution of short-term measurements and annual average radon concentrations.

(5) Given the joint probability distribution developed above, EPA modeled the outcome of subsequent tests by developing conditional probability distributions based on the annual averages in homes that would be required to retest.

(6) EPA determined the overall misclassification rates for each option by summing the probabilities of misclassification under all possible outcomes for the option.

Step 1 — EPA's first step was to develop a statistical model that could explain how short-term measurements could deviate from annual averages. EPA's model is presented in Equation 1. After starting with the annual average as an expected value for a short-term measurement, EPA's model includes components that could contribute to deviations from annual averages. The model includes components for three error types — measurement error, temporal error, and spatial error — as well as the bias of radon concentration on the testing floor level, compared to the overall house annual average. EPA developed upper and lower bound estimates for all three error types used in its model.

$$\begin{matrix} \textit{Lowest Living Level} \\ \textit{Short-Term} \\ \textit{Measurement} \end{matrix} = \left(\begin{matrix} \textit{Annual} \\ \textit{Average} \end{matrix} \times \begin{matrix} \textit{Lowest Living Level} \\ \textit{Floor Bias} \end{matrix} \right) \times \left(e_{measurement} \times e_{temporal} \times e_{spatial} \right) \qquad (1)$$

An **annual average** is the mean radon level over time for a particular house. It reflects an average of the radon concentrations in all lived-in levels of a home. To calculate the misclassifications associated with each option, EPA needed to know the distribution of annual averages, since these determine what short-term measurements will be observed. For its analysis, EPA assumed that nationwide annual average radon levels had the distribution suggested by the Agency's National Residential Radon Survey (U.S. EPA/Office of Radiation Programs 1991a) for homes that EPA recommends test.

Measurement error is the error associated with either a testing device itself or in the reading of an exposed device by a laboratory technician. EPA's analysis covered only two types of measurement devices, charcoal canisters and alpha track detectors. Other commonly used measurement devices, such as electrets and continuous monitors, would perform similarly, if not better. If the latter were the case, then the misclassification rates listed in Exhibit 3-5 would be reduced slightly. A description of commonly used measurement devices is included in Appendix C.

For its analysis, EPA assumed that there was an equal likelihood of using charcoal canisters and alpha track detectors. Initially, EPA intended to model the measurement device error associated with charcoal canisters and alpha track detectors separately. For this purpose, EPA obtained data on charcoal canisters from EPA/State surveys (U.S. EPA/Radon Division 1990c) and data on alpha track detectors from a New York State survey commissioned by the State government (Perritt 1990). The EPA/State surveys provided paired charcoal canister and alpha track detector data on over 1,000 homes. The New York State survey had information on over 2,000 homes. After comparing these two sources of data, EPA concluded that no significant difference in measurement error existed between these two types of devices. EPA data show that typical measurement device error for both charcoal canisters and alpha track detectors is approximately 15 percent in the lab and 30 percent in the field, in the vicinity of 4 pCi/L (U.S. EPA/National Air and Radiation Laboratory 1991). The percentage measurement error is higher at lower radon levels, and lower at higher radon levels.

Temporal error is the error associated with the time of year a short-term measurement was taken. EPA assumes that the annual average is constant, but that within any day, month, or year there will be fluctuations in the radon level. At any time in the year, therefore, the exact relationship of a short-term measurement to the annual average will not be known exactly. This inexactness is the temporal error.

Since there are seasonal differences in the radon level of a home, the false positive rate could be greater in the winter than in the summer, or the false negative rate could be greater in the summer than in the winter (Marcinowski 1990; Condon et al. 1990). However, Radon Program Experience indicates that measurements are currently being conducted during all seasons of the year. This indicated the need for a testing protocol that could be used throughout the entire year, so the analysis focused on testing occurring equally during all seasons.[3]

Research indicates that temporal errors are the biggest contributor to the overall error associated with short-term measurements. EPA simulated charcoal canister and alpha track device measurements separately in its analysis because of the difference in temporal errors associated with

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[3]It is unreasonable to assume that closed-house conditions would be maintained for a 90-day test. Because there is a lack of data on compliance with closed-house conditions, EPA assumed both open-house and closed-house conditions would occur with equal frequency for testing periods of this duration.

these devices due to their different recommended durations of exposure. Because of the significance of seasonal variation in radon levels, EPA estimates that a measurement is subject to a temporal error ranging from as low as 40 percent, for certain tests made under closed-house conditions using alpha track detectors, to as high as 80 percent, for certain tests made under open-house conditions using charcoal canisters. These estimates have been based on an EPA study comparing short-term measurements to annual averages (U.S. EPA/Radon Division 1990d). Because data in this study are from a cold climate area with a long heating season and since the seasonal variability is greater in areas with long heating seasons, the estimate of temporal error is conservative (i.e., high). Chmelynski (1992) provides more detail on the methods used to estimate temporal error.

Spatial error is the error associated with the relationship between the radon concentration on the lowest lived-in floor, where short-term tests are made, and the overall radon concentration in a house. Spatial error does not account for the entire difference between the radon level on a particular floor and the overall home average, but only the unpredictable component of this difference. EPA estimates that short-term measurements are subject to spatial error that is about 40 percent of a measurement. This estimate has been based on data gathered for a study comparing basement radon levels to first floor radon levels (Ronca-Battista 1989).

Since annual averages in most cases take into account the radon levels on more than one floor within a house, short-term measurements may be inaccurate if the floor on which they are taken does not tend to have the same radon level as the home as a whole. EPA identified the predictable difference between the radon level on a given floor and the overall home average as the **floor bias**. When radon measurements (short-term or long-term) on all lived-in levels in a home are considered together, the resulting aggregate has no floor bias, since it would be expected to approach the house overall average. Houses with only one lived-in level would not have any floor bias, since the lived-in level average would be the same as the overall house average.

Research indicates that for 75 percent of homes, the lowest lived-in level is the first floor (Sterling et al. 1985). Research also indicates that for most homes, the first floor average is roughly reflective of the overall home annual average (Peake 1990). Accordingly, the Agency assumed for its modeling that there would be no floor bias associated with lowest lived-in level measurements.

The components of Equation 1 are multiplicative (reflecting how lowest lived-in floor bias and error components affect the annual average) rather than additive. To make this equation more conducive to statistical analysis, EPA applied a logarithmic transformation to both sides of the equation. Equation 2 shows that the log of a single short-term measurement can be modeled using five additive components.

$$\log\begin{pmatrix}\textit{Lowest Living Level}\\ \textit{Short-Term}\\ \textit{Measurement}\end{pmatrix} = \left[\log\begin{pmatrix}\textit{Annual}\\ \textit{Average}\end{pmatrix} + \log\begin{pmatrix}\textit{Lowest Living Level}\\ \textit{Floor Bias}\end{pmatrix}\right] + \left[\log e_{measurement} + \log e_{temporal} + \log e_{spatial}\right] \qquad (2)$$

Steps 2 and 3 — To make its calculation of the overall misclassification rates for an option, EPA needed to calculate separately the misclassification rates associated with each possible outcome of the option (i.e., what type of confirmatory test, if any, would be required based on the measurement observed in the initial test). In Exhibit 3-8, the different outcomes possible under each option are graphically separated into different boxes. The types of error associated with each possible outcome are also indicated. In this exhibit, unshaded areas for each option indicate the range of initial short-term measurements for which no confirmatory tests are required. Shading indicates areas where errors are dependent on the combination of initial short-term and confirmatory tests. Option E is completely unshaded, unlike other options, because for this option no confirmatory tests are performed.

Step 4 — For each option, EPA began its analysis of misclassification rates by focusing on the possible outcomes of the initial test. Based on the statistical model shown in Equation 2, EPA developed a joint probability distribution of annual average radon concentrations and expected short-term measurements. A joint probability distribution indicates the probability with which certain short-term measurements can be expected to be observed at a given annual average. EPA needed to develop a joint probability distribution of annual averages and short-term measurements to determine the proportion of homes in which a confirmatory test would be required under each option, assuming the national distribution of annual average radon levels.

Because a joint probability distribution indicates the distribution of short-term measurements that will occur for a given distribution of annual averages, it can be used to calculate the misclassification rates attributed to testing outcomes not requiring a confirmatory test. Options A, B, C, and F all require one initial short-term test with an action level of 4 pCi/L. Exhibit 3-8 indicates that the initial short-term test components in these options will produce false negatives in the same way, since all four options have one initial measurement and the same action level. The overall misclassification rates of these four options vary due to differences in the confirmatory test required when the initial measurement is above the action level. Since Option D requires further testing if the initial measurement is above 2 pCi/L, this option has fewer false negatives associated with the initial testing. Option E, unlike other options, bases mitigation decisions on the average of several initial short-term tests. This option requires no further testing regardless of the radon level observed using short-term testing. Consequently, Steps 5 and 6 are not required for this option.

Step 5 — Using the joint probability distribution of annual averages and short-term testing that it developed for Options A, B, C, D, and F, EPA developed conditional probability distributions to model the outcome of the possible subsequent tests required under each option. Conditional probability distributions indicate the expected frequency of results considering only the subset of a population that has already met a given condition.

In EPA's analysis, the subset of the population being considered was only those homes that had short-term measurements above the action level (Options A, B, C, and F) or the trigger level (Option D). For each possible outcome in an option, EPA developed a separate conditional probability distribution. In all cases, the conditional distributions took into account the subpopulation required to retest and whether a long-term or short-term test was required. As part of its analysis of conditional distributions, EPA focused on the probabilities of false positives occurring at homes with annual average radon levels between 2 pCi/L and 4 pCi/L. If these homes are mitigated, it is likely that the radon levels in them will decrease and the occupants will receive some benefits from lowering their exposure to radon.

EXHIBIT 3-8
COMPARISON OF FALSE NEGATIVES AND FALSE POSITIVES UNDER RADON TESTING OPTIONS THAT EPA CONSIDERED

4 pCi/L Action Level For Options A-C. Initial Test Above This Level Leads to Confirmatory Testing.

Option A

If ST test is below action level, no retest is required. False negatives may result.	*LT test is assumed to result in no misclassifications.*

Option B

If ST test is below action level, no retest is required. False negatives may result.	*ST confirmatory test will result in false positives and false negatives.*

Option C

If ST test is below action level, no retest is required. False negatives may result.	*9% - LT (all correct)* *91% - ST (false negatives and false positives.)*

Option D

2 pCi/L Trigger Level — *10 pCi/L Trigger Level*

If ST is below 2 pCi/L, no retest is required. False negatives may result.	*No misclassification resulting from confirmatory LT.*	*ST required. False positives will result.*

Option E

If average of ST measurements is below action level, false negatives may result.	*If average of ST measurements is above action level, false positives may result.*

Option F

If ST test is below action level, no retest is required. False negatives may result.	*ST confirmatory test will result in false positives and false negatives.*

4 pCi/L Action Level

ST = Short-Term Test
LT = Long-Term Test

Note: Shaded areas indicate results dependent on both initial and confirmatory tests.

Step 6 — To determine the overall misclassification rates for each option, EPA summed probabilities in the joint probability distribution of annual averages and initial short-term tests and in the conditional probability distributions of houses required to retest. To calculate the total of false negatives, EPA summed probabilities that a testing outcome would classify a home as being below the action level when the annual average was actually above the action level. To calculate the total of false positives, EPA summed probabilities that a testing outcome would classify a home as being above the action level when the annual average was actually below the action level. EPA aggregated false negatives and false positives for possible outcomes under each option to generate the results presented in Exhibit 3-5. Because EPA's analysis in essence is a comparison between the accuracy of the option relying on a confirmatory long-term test and the accuracies of options relying to varying degrees on confirmatory short-term tests, the Agency simplified its analysis by assuming that long-term tests did not result in any misclassifications. This is because temporal variation is the largest contributor to the misclassification rate and for year-long tests, this error term is negligible. The misclassification for those options utilizing a long-term confirmatory test is insignificant and is represented as resulting in no misclassifications in the results.

Section V: Analysis of Options

Options A, B, C, and F begin with a single short-term test, followed by a confirmatory long-term or short-term test if the result of the initial test is above the action level. Because these four options begin with the same procedure, they share the same rate of false negatives among homes not requiring a subsequent test. The difference in misclassification rates among these four options is due strictly to differences in the procedures they use for follow-up testing. To determine the misclassification rates associated with follow-up tests under each option, EPA developed conditional probability distributions for the subset of homes expected to need confirmatory testing.

Like Options A, B, C, and F, Option D also begins with a single short-term test. Option D, however, has a trigger level of 2 pCi/L, and an additional trigger level at 10 pCi/L that is discussed further below in the documentation of EPA's analysis of Option D. Option D's lower trigger level of 2 pCi/L for initiating confirmatory testing leads to a lower rate of false negatives than Options A through C since homes with measurements between 2 to 4 pCi/L are subject to long-term tests. Only false positives happen in measurements above 10 pCi/L, since in the interval of 2 to 10 pCi/L long-term testing is done.

Option E, unlike other testing alternatives, consists of multiple initial tests and does not require confirmatory testing. Accordingly, for this option EPA analyzed only the misclassification rates due to initial testing. EPA's approaches for determining the misclassification rates for the six options are discussed in more detail below.

OPTION A — This alternative recommends a confirmatory long-term test whenever the results of the short-term measurement are above the action level. As discussed above, the misclassifications for long-term measurements are insignificant and, therefore, all long-term tests were represented as resulting in correct classifications of homes concerning the need for mitigation. Accordingly, to compute the misclassification rates associated with this option, EPA needed only to consider the probabilities that false negatives would occur with the initial short-term test. These probabilities were obtained from a joint probability distribution of annual averages and expected results of the initial short-term test. This joint probability distribution was based on Equation 2, which states the assumed relationship between short-term measurements and long-term annual averages. See Steps 1 through 3 of Exhibit 3-9 for the derivation of the joint distribution.

OPTION B — This option is similar to Option A, except that it requires a confirmatory short-term test when the initial short-term measurement exceeds 4 pCi/L. Some false negatives result from this testing procedure when the initial short-term measurement is below the action level. Both false negatives and false positives are associated with the option when the initial short-term measurement is above 4 pCi/L, depending on the outcome of the second test. Exhibit 3-9 illustrates the key steps in determining false positives for Option B.

To analyze the misclassification rates associated with Option B, EPA separately considered the range of initial short-term measurements below the action level and above the action level. For its analysis of Option A, it already had analyzed the probability of obtaining false negatives associated with short-term measurements below 4 pCi/L.

To calculate the misclassification rates associated with the average of two short-term measurements with a 30-day waiting period between them, EPA modified its statistical model. When EPA was analyzing the misclassifications associated with only one short-term test, as in Option A, it took into account the temporal error associated with a radon measurement made at <u>one</u> given time of the year. If a second short-term measurement were made at the same time as the first (as in the case of Option E, described below), the Agency would still be dealing with the same amount of temporal error. In statistics, the strength of the relationship between sequential samples is called serial correlation. In the absence of serial correlation, the averaging of two measurements would lead to a $1/\sqrt{2}$ reduction of the temporal error associated with short-term measurements, and hence to a large reduction in the misclassification rate. To determine the degree to which a second sample would reduce temporal error, EPA needed to take serial correlation into account in its statistical model. In considering Option B, the Agency did this at a point in its analysis corresponding to Step 5 in Exhibit 3-7.

To determine how much serial correlation there is between two short-term measurements, EPA studied the variations in radon levels in homes over time in the course of its Butte, Montana study (U.S. EPA/Radon Division, 1990d). The reduction in uncertainty obtained using a second, serially correlated short-term measurement was determined by computing the serial correlation of various short-term averages separated by a set number of days using data on daily radon levels in these homes. When the short-term measurements are made under closed-house conditions, a serial correlation of approximately 0.4 was obtained for 2- to 7-day tests separated by 30 days. For 90-day tests separated by 90 days, the serial correlation was lower, approximately 0.1. Variance reduction factors were then computed, and these factors were then applied to the temporal error term in Equation 2. The reduction in temporal error due to seasonal variation resulting from a second, uncorrelated measurement would be approximately 30 percent ($1 - [1/\sqrt{2}]$). The estimated reduction in temporal error for 2- to 7-day tests separated by 30 days was 16 percent, compared to 26 percent for back-to-back 90-day tests.

After estimating the amount of reduction in uncertainty that a confirmatory short-term measurement would bring, EPA developed a conditional probability distribution of the average of two sequential short-term tests among houses with an initial short-term test above 4 pCi/L. Evaluation of the required conditional distribution was based on the joint (trivariate) distribution of two serially correlated short-term tests and the long-term annual average, as described in Step 5 of Exhibit 3-9.

EXHIBIT 3-9

HOW FALSE POSITIVE MISCLASSIFICATION RATES WERE CALCULATED FOR OPTIONS

Step (1) — EPA Developed a Model for Expressing Short-Term Measurements in Terms of the Long-Term Annual Average and Several Components Contributing to Deviations from this Average (See Equation 2 in Text)

$$\log\begin{pmatrix}\textit{Lowest Living Level}\\ \textit{Short-Term}\\ \textit{Measurement}\end{pmatrix} = \left[\log\begin{pmatrix}\textit{Annual}\\ \textit{Average}\end{pmatrix} + \log\begin{pmatrix}\textit{Lowest Living Level}\\ \textit{Floor Bias}\end{pmatrix}\right] + \left[\log e_{measurement} + \log e_{temporal} + \log e_{spatial}\right]$$

Step (2) — EPA Assumed Values or Ranges of Values for All Additive Components Based on EPA Studies

Lowest Living Floor Bias — none
Measurement Error — 30% at 4 pCi/L
Temporal Error — 40% to 80%
Spatial Error — 40%

Step (3) — EPA Developed a Joint Probability Distribution to Predict the Homes Requiring Confirmatory Testing. The model in Step (1) estimates the conditional distribution of ST_1 given the annual average, $f(ST_1,AA)$.

$$\textit{Probability}\ \{ST_1 > 4\,pCi/L,\ AA < 4\,pCi/L\} = \int_{ST_1=4}^{\infty}\int_{AA=0}^{4} f(ST_1, AA)\ dST_1\ dAA$$

where ST_1 = *Initial Short-Term Measurement,* AA = *Annual Average*

Step (4) — EPA Analyzed the Percentages of Initial Short-Term Measurements Above 4 pCi/L for Certain Ranges of Annual Averages (percentages are expressed as proportion of homes testing)

True Annual Average Range	Short-Term Tests Above 4 pCi/L
0 - 1 pCi/L	1.9%
1 - 2 pCi/L	2.4%
2 - 3 pCi/L	1.9%
3 - 4 pCi/L	1.6%

Step (5) — EPA Developed a Conditional Probability Distribution for the Average of Two Short-Term Tests, Taking into Account Serial Correlation, Given the Distribution of Annual Averages and the Initial Measurement

$$\textit{Prob}\ \{ST_{AVG} > 4\,pCi/L \mid ST_1 > 4\,pCi/L,\ AA < 4\,pCi/L\} = \frac{\int_{AA=0}^{4}\int_{ST_1=4}^{\infty}\int_{ST_2=8-ST_1}^{\infty} f(AA, ST_1, ST_2)\ dAA\ dST_1\ dST_2}{\int_{AA=0}^{4}\int_{ST_1=4}^{\infty} f(AA, ST_1)\ dAA\ dST_1}$$

where ST_1, ST_2 = *Short-Term Measurements,* AA = *Annual Average*

Step (6) — EPA Estimated the Percentage of Short-Term Measurements above 4 pCi/L and Summed Them to Get Total False Positives

True Annual Average Range	Option B False Positives
0 - 1 pCi/L	0.4%
1 - 2 pCi/L	0.9%
2 - 3 pCi/L	1.1%
3 - 4 pCi/L	1.2%
0 - 4 pCi/L	3.6%

To obtain the overall misclassification rate for Option B, EPA aggregated the probabilities of obtaining false negatives when follow-up testing was not required with the probabilities of obtaining false negatives and false positives when a confirmatory test was required. (In addition to leading to a false positive by confirming an initial short-term test, the confirmatory test can lead to a false negative if it does not agree with an initial test when the radon level is actually high.) These results are presented in Exhibit 3-5.

OPTION C — To analyze the misclassification rates associated with Option C, EPA viewed this testing procedure as a combination of Options A and B. Those options, respectively, require initial short-term measurements to be followed by a confirmatory long-term or short-term measurement.

Since Option C allows individuals to use either a confirmatory long-term or short-term test, EPA's first task in analyzing the misclassification rates associated with this option was to determine the frequencies with which each type of test would be used. EPA relied on a study it had done of the public's willingness to use long-term tests. This study indicated that only 9 percent of the public was willing to use long-term tests (Johnson 1990). Accordingly, EPA assumed for its analysis of Option C that for initial short-term measurements that were above the action level, 9 percent would be followed by confirmatory long-term measurements and that the remaining 91 percent would be followed by confirmatory short-term tests.

Once EPA had assumed how the public would respond to the flexibility in the testing procedure offered under Option C, it could determine the misclassification rate of the option by combining its analytical approaches for Options A and B. The Agency weighted the results of these two options with factors of 0.1 and 0.9, respectively, to simulate the choice of long-term and short-term testing procedures by the public. The results are presented in Exhibit 3-5.

OPTION D — Like Option C, this option can be viewed for analytical purposes as a combination of Options A and B. Option D, however, does not give the public the choice between long-term and short-term tests, but rather specifies which type of confirmatory test will be used, based on the concentration of the initial short-term measurement. For measurements between 2 pCi/L and 10 pCi/L, a confirmatory long-term test is required. For measurements above 10 pCi/L, only a confirmatory short-term test is required. Of course, for measurements below 2 pCi/L, no confirmatory test is required at all.

To determine the total amount of misclassification associated with Option D, EPA considered the three possible outcomes of the initial short-term test separately. Exhibit 3-8 illustrates what types of misclassification errors are associated with these three outcomes. Because initial short-term measurements below the lower trigger level of 2 pCi/L do not include any scenarios leading to mitigation, some false negatives occur. Roughly 93 percent of all homes in the population considered did not require a confirmatory measurement. Initial short-term measurements above the lower trigger level (2 pCi/L), but below the higher trigger level (10 pCi/L), were represented as not resulting in any misclassifications at all, since the misclassifications for long-term measurements were insignificant. Roughly 4 percent of all homes that should test had this outcome. Results above the higher trigger level always indicated that mitigation was necessary, since the average of the two short-term tests was always at least 5 pCi/L. In practice, however, if there is a large disparity between the results of the first and second test, additional evaluation may be necessary to determine the need for mitigation.

Less than 3 percent of all homes in the population had initial short-term tests with measurements above 10 pCi/L.

Based on its analysis of Options A through C, EPA had already obtained a joint probability distribution of annual averages and initial short-term measurements. From this joint probability distribution, EPA determined the probability of false negatives due to initial short-term tests incorrectly providing measurements of less than 2 pCi/L.

Since EPA assumed that no misclassifications would be associated with long-term measurements, it determined that all initial short-term measurements at least equal to 2 pCi/L but below 10 pCi/L would result in correct final classifications, regardless of the accuracy of the initial short-term test.

EPA knew that all initial short-term measurements above 10 pCi/L would require mitigation. To determine the number of false positives associated with this outcome, EPA aggregated the probabilities of having a false positive above 10 pCi/L from the joint probability distribution. (EPA did not have to take into account the serial correlation between the two short-term tests under this option, since it assumed that the outcome would always result in an assessment that mitigation was necessary.)

To determine the overall misclassification rate associated with Option D, EPA combined the probabilities that false negatives would occur due to initial short-term measurements below 2 pCi/L with the probabilities that false positives would occur due to initial short-term measurements above 10 pCi/L. The results are presented in Exhibit 3-5.

OPTION E — Unlike Options A through D, Option E requires that short-term tests be conducted on all levels of a home that are lived-in levels (i.e., floors on which individuals spend more than 4 hours of their time). Because this testing procedure would result in a number of different testing patterns based on home type and lifestyle, EPA repeated its analysis of this option for three different housing situations that could occur. The summarized results presented in Exhibit 3-5 reflect a weighted average of the misclassifications for the different housing situations. The first situation EPA considered was an individual living on one level. In this case, the individual would make two tests on the same level, in different locations. The second situation EPA considered was an individual living on two levels, either basement and first floor or first floor and second floor. The third situation EPA considered was an individual living on three levels.

After the Agency had determined the reduction in error components in Equation 1 under all three scenarios, EPA derived three separate joint probability distributions for this option. All of these distributions differed slightly from the single short-term measurement/annual average joint probability distribution developed for Options A through D. Because each of the three scenarios had a different joint probability distribution, each also had a different misclassification rate.

OPTION F — Like Option B, this option required a confirmatory short-term test when the initial short-term measurement exceeded 4 pCi/L. As Exhibit 3-8 illustrates, some false negatives resulted from this testing procedure when the initial short-term measurement was below the action level. Both false negatives and false positives were associated with the option when the initial short-term measurement was above 4 pCi/L, depending on the outcome of the second test. Unlike Option B, however, this option does not average test results.

To analyze the misclassification rates associated with Option F, EPA modified the approach that it had taken for the analysis of Option B. After estimating the amount of reduction in uncertainty that a confirmatory short-term measurement would bring given the serial correlation between two sequential measurements, EPA developed a conditional probability distribution of the probability of outcomes for a second short-term among houses with an initial short-term test above 4 pCi/L. To obtain the overall misclassification rate for Option F, EPA aggregated the probabilities of obtaining false negatives when follow-up testing was not required with the probabilities of obtaining false negatives and false positives when a confirmatory test was required. (In addition to leading to a false positive by confirming an initial short-term test, the confirmatory test could lead to false negatives if it does not agree with an initial test when the radon level is actually high.)

Section VI: General Conclusions

Analysis of options that rely on short-term tests to decide whether to mitigate existing radon levels shows that these tests can be used effectively. For the action level of 4 pCi/L, the Agency found that short-term testing procedures can lead to a reasonably low level of misclassification of homes with respect to the action level. EPA found that the skewed distribution of radon levels towards the lower end of the distribution served as a strong countervailing factor to any misclassifications created by short-term testing errors. The results are based on examination of the most commonly used radon measurement devices and consideration of testing throughout the year.

Option A, which requires a single short-term measurement followed by a confirmatory long-term test, parallels the testing procedure recommended in the original *Citizen's Guide*, except that testing would be conducted on the lowest lived-in level. Under Option A, 98 percent of homes were correctly classified with regard to the need for mitigation. To determine how testing procedures relying on a short-term confirmatory test would compare to this option, EPA developed and analyzed five other options (Options B through F). EPA found that all of these options produced results yielding at least 94 percent correct classifications. Similarly, each option has the potential to save a significant number of lives, although there is some variation in the potential number of lives saved and the number of lives not saved under each testing procedure.

Given these results, the Agency determined that on the basis of misclassification, none of the options should be rejected. EPA decided that other factors, such as procedure simplicity, should be weighed with the trade-off of increased levels of proper classification in making the decision on the testing option to recommend in the revised *Citizen's Guide*. As stated previously, current experience indicates that few people are actually taking long-term follow-up tests, and that most people who mitigate do so based on a single short-term measurement. This is supported by risk communication research that indicates that only 9 percent of the population is willing to conduct annual tests (see Chapter 6 for a more detailed discussion). This suggests that a testing protocol that relies exclusively on long-term confirmatory tests is unlikely to be followed by most of the public. Therefore, although Options A and D have more desirable error rates than other options, it is likely that compliance with these options would be low. Selection of either of these two options is apt to result in a situation in which decisions are most often made based on a single short-term measurement when homeowners do not complete the testing process. By explicitly providing homeowners a process that calls for more than one short-term test, all of the other options considered would be more likely to result in people having better information to use in reaching a mitigation decision.

On the other hand, since long-term measurements are more desirable than short-term measurements and since there are some individuals who are willing to conduct long-term tests, the use of long-term tests should not be precluded from the testing protocol. As a result, Options B, E, and F, which do not include long-term tests, are also undesirable.

Option C offers an effective compromise between these different approaches. It promotes long-term testing by people who are willing to conduct long-term tests and recommends an effective short-term test as an alternative for people who are not willing to conduct a long-term test. Therefore, consistent with the advice from EPA's SAB, the revised *Guide* recommends Option C because it should maximize the total risk reduction the public would gain through future testing and mitigation.

CHAPTER 4

MITIGATION TECHNOLOGY

This chapter explains different types of available and emerging radon mitigation techniques, their effectiveness, and costs. The chapter is based primarily on research that EPA's Office of Research and Development (ORD) has conducted and results from a survey of commercial radon mitigators that EPA's Office of Radiation Programs (ORP) conducted in 1989.

> THE AIMS OF THIS CHAPTER
>
> (1) To describe different radon mitigation techniques.
>
> (2) To present information on the effectiveness and cost of different techniques.

Section I: Mitigation Methods

Preventing radon entry and reducing radon concentrations after entry are two common mitigation strategies. Preventing entry is often the best strategy since it has a high probability of success, even in locations with very high radon levels. Techniques that reduce radon after entry are most appropriate for buildings with relatively low radon levels, where radon entry cannot be prevented, or in which increased ventilation could provide valuable benefits in addition to radon reduction.

Regardless of the strategy considered, any mitigation plan must take into account a number of considerations. In addition to controlling radon, the mitigation system should be unobtrusive, quiet, and capable of indicating system failure. It should be economical and easy to maintain and operate. Mitigation systems also must be a permanent part of the building rather than portable or window-mounted devices that can be removed when the building is sold.[1]

Methods that Prevent Radon Entry

Gaseous elements contained beneath the earth's surface can migrate through openings in soil or rock and be released from the ground, depending on a number of factors. Soil gas is drawn indoors by the differential between relatively low air pressure in a house and the higher air pressure in the soil. **Depressurization** of the surrounding soil and foundation can prevent radon-bearing soil gas from entering a house. Experience also shows that depressurization may be made more effective by sealing cracks and other openings in floors and walls. Sub-slab depressurization is the preferred approach because of its effectiveness, applicability, and ease of installation. Whenever any active system is installed, the mitigator should test the house for combustion backdrafting.

Sub-slab depressurization (also called sub-slab suction) is a popular mitigation technology. Once soil gas accumulates in the soil and aggregate underlying the concrete slab in a basement or slab-on-grade house, it can migrate indoors through any openings in the slab. Active sub-slab depressurization systems use pipes running down through the slab and up to a fan that vents the gas

[1]According to the Interim Mitigation Standards for the EPA Radon Contractor Proficiency Program.

outdoors. The fan reduces air pressure in the soil, thereby reversing the pressure gradient and causing air to flow out of the house and into the soil through any openings in the slab. Passive sub-slab depressurization systems also can be used, but are less effective than active sub-slab depressurization. Soil depressurization under a plastic or rubber membrane, or **sub-membrane depressurization**, is used in buildings with earth-floored crawlspaces or basements (Hubbard et al. 1987). Its function is essentially the same as sub-slab depressurization, except that the mitigator constructs a barrier as a collection cover rather than using the existing slab as a barrier.

If a perforated drain tile is present, **drain-tile depressurization** can be used to create a negative pressure field surrounding the foundation walls to draw soil gas away from possible entry routes. **Block-wall depressurization** consists of using a fan and duct work to draw suction on the hollow interior cavities of a concrete block wall. By maintaining a lower pressure in the void network within the block wall than in the basement, this technique forces the flow of soil gas to be outward rather than into the basement. In a **baseboard depressurization** system, a "baseboard" is installed around the entire perimeter of the basement and fans are used to depressurize the block wall cavities. This system evenly distributes the pressure field and may simultaneously handle wet foundation problems (Fowler et al. 1988; Henschel 1987).

Sub-slab and block-wall pressurization have been tested as alternatives to depressurization in a number of government research projects. Essentially, these techniques reverse the fan in a soil depressurization system, creating a positive shield of diluted air around a building. These methods have had mixed results (Henschel and Scott 1987; Hubbard et al. 1987; Pyle et al. 1988; Turk et al. 1986). In certain conditions, they perform better than soil depressurization. (The selection of these techniques requires caution when pesticides have been applied nearby.)

Basement pressurization prevents soil gas from entering the building and reduces radon concentrations by dilution (Nitschke et al. 1985; Turk et al. 1986). It can be accomplished by using a fan to pull indoor air from the upstairs lived-in area and blow it into the basement. This process creates a higher air pressure in the basement than outside, causing indoor air to flow through cracks and holes in the basement out into the soil. Basement pressurization should be considered an alternative for houses that are not compatible with soil depressurization techniques. Adequate long-term data are not available to determine the effect of basement pressurization on energy costs or whether condensation will occur as basement air is forced out through cracks above grade.

Sealing openings to the soil is helpful as a supplementary activity to other methods as it closes radon entry pathways and can improve the extension of depressurization on the soil. However, the results of sealing are unpredictable when it is used alone as the primary mitigation method. Success seems to be based on the number of cracks and holes in the slab or foundation, as well as their connection to radon transport mechanisms. If the house is situated in a permeable soil with elevated radon concentrations in the soil gas, it is very difficult to make a sealing-only technique work. If there is only one primary transport mechanism connected to a single entry point, sealing could make a difference, assuming the entry route could be sealed.

Isolation is another form of sealing that can be used to prevent radon entry. This technique closes the openings between a large area of the substructure and the adjacent lived-in space, thereby isolating exposed soil and rock under, around, or within a house that can be both a major source and entry route for radon (Nitschke et al. 1985; Turk et al. 1986). When used in combination with ventilation, isolation can be a very effective mitigation approach. Unfortunately, many spaces cannot be easily isolated from the lived-in space and ventilated without causing winter freezing problems or summer moisture problems. Crawlspaces lend themselves to isolation and ventilation, unless they contain air distribution duct work that makes isolation difficult. Lived-in space radon concentrations can be reduced by passively ventilating a crawlspace, depressurizing the crawlspace, pressurizing the crawlspace, or creating a balanced flow. The two primary goals are to (1) decouple the lived-in area from the soil, and (2) reduce concentrations in the crawlspace by dilution.

Radon entry also can be deterred by **reducing negative pressure** in the lowest level of the house, since this pressure is the main driving force for infiltrating soil air. Negative pressure can be reduced by sealing thermal bypasses at the ceiling level or supplying makeup (combustion or exhaust) air to appliances such as furnaces, boilers, and bath fans that depressurize a house by using and exhausting air to the outside. Currently, reducing negative pressure is not a primary mitigation technique, but is used to enhance the performance of other mitigation efforts or to prevent a radon problem in new construction (very little research has been done to evaluate it as a primary mitigation approach). This technique also may reduce a home's heating needs.

Methods that Remove Radon from a Building

Once radon has entered a building, a variety of mitigation techniques can be used to reduce the radon concentrations. Any radon problem can be reduced to some extent by blowing air through the house. However, the use of increased ventilation is usually limited to homes in mild climates with less than 20 pCi/L of radon because of the energy costs and comfort problems that would be associated with the addition of large amounts of ventilation air. Best results are obtained when basement areas are ventilated. Ventilation also can provide additional benefits such as the dilution of other indoor air contaminants.

Mechanically powered ventilation is used either to blow air into the house or to supply and exhaust air simultaneously. The indoor air pressure is increased (i.e., negative pressure is reduced) if air is blown into the house, possibly leading to higher radon reductions than would be expected from dilution alone. However, in climates where heating is necessary, blowing unconditioned air into the house also would tend to increase energy usage and may increase the risk of moisture problems in the building shell. If one fan is blowing in the same amount of air that another fan is removing, the indoor pressure level will not be affected. Therefore, if other factors such as wind and temperature remain constant, the influx of radon will remain the same and the radon concentration will go down in proportion to the increase in the ventilation rate. If mechanically powered ventilation is used only to exhaust air from the house, radon may be increased due to the negative pressure induced inside (Hubbard et al. 1987; Lencheck et al. 1987).

Ventilation systems can be designed with or without a heat recovery system. A typical system without heat recovery can employ a duct that draws outdoor air into the cold-air return of a forced-air furnace when the furnace is operating (Pyle et al. 1988). When the furnace fan is not operating, fresh air enters only in response to pressure differentials between outside and inside air, thereby reducing pressure differentials. The furnace heats this air and distributes it to the house. This small amount of air will probably not move the house from negative pressure to positive pressure, but it will reduce the negative pressure in the basement.

Heat recovery ventilators (HRVs or air-to-air heat exchangers) are designed to bring in the same amount of air that the system is exhausting. The incoming air stream is preheated by the outgoing airstream (Fowler et al. 1988; Henschel and Scott 1987; Lenchek et al. 1987; Nitschke et al. 1988). This process reduces the energy costs and associated discomfort. Because the balanced flow should have no impact on basement pressure, radon gas reduction will occur only through dilution. The radon concentration will be reduced by the inverse of the ventilation increase. If a basement is large, leaky, or has greatly elevated radon levels, it will be very difficult to make a significant difference with ventilation.

Another radon removal device circulates interior air through alternating **charcoal filtration beds**. As interior air circulates through one bed that accumulates radon and its decay products, exterior air is circulated through the second bed and exhausted outside. Thus, the beds collect radon and its decay products from the indoor air which are then transported outside by the flow of outside air. The airflows alternate between beds to maintain radon removal efficiency.

Passive ventilation is achieved by opening windows while ensuring that the house is not depressurized. Passive ventilation works by diluting the indoor radon concentration and by neutralizing the pressure differentials between the inside and the outside. In one study, by opening all the windows the leakage area of the house was increased by a factor of 10 to 20, and radon concentrations in the house were reduced by a factor of 4 to 10 (Hubbard et al. 1987). Because this method can be used only when the outside weather permits, it is not an adequate permanent solution.

<u>Other Methods under Development</u>

There are several other methods that have not yet been shown to reduce radon health risks, but which are still the subject of research and may prove to be significant after further improvement. These alternate methods include air cleaners and space ionization. EPA does not recommend the use of these techniques because they can be easily removed from the structure.

Air cleaners that are commonly used to remove airborne particulates and condition indoor air undoubtedly remove some radon decay products. However, many questions remain concerning the relative health effects of the decay products that are not removed (the unattached fraction).

Space ionization removes radon decay products from indoor air by ionizing and circulating the air. Many units may be needed to treat an entire house effectively. Also, no easy way currently exists to monitor the performance of units that remove decay products without affecting radon gas concentrations. Use of these devices should be considered only where preventative techniques are not feasible, or as a temporary technique until other controls are implemented.

Section II: Effectiveness and Cost

This section summarizes the effectiveness and cost of various radon mitigation techniques. Effectiveness is measured by the reduction in initial radon levels, and cost includes all the associated costs of employing each technique. There are two primary sources of information on mitigation effectiveness and cost: EPA research and demonstration projects and the Private Sector Radon Mitigator Survey (U.S. EPA/Radon Division 1990a). Each one of the sources is summarized individually. (The Office of Radiation Programs also has constructed a mitigation cost model that uses the information summarized here along with additional information supplied by private mitigators to estimate costs. That model is explained in Chapter 5.)

Effectiveness

Several research projects on the reduction in radon levels achieved by various mitigation techniques have been conducted by EPA's ORD. Pre- and post-mitigation radon concentrations were measured in the basement for houses with basements, and in the lived-in area for houses with slab-on-grade and crawlspaces. Various measurement methods were used in the different projects (e.g., alpha track detectors, charcoal canisters, continuous monitors). ORD's program was designed to reduce radon levels in homes with very high initial radon levels and often was conducted in houses that had been identified as difficult to fix.

Through this research, ORD has found that radon levels in almost all homes can be reduced to less than 4 pCi/L (U.S. EPA/ORD 1989). In fact, radon levels very often were reduced to less than 2 pCi/L, even though the mitigation efforts were only trying to reach an action level of 4 pCi/L. EPA believes that radon reduction to 2 pCi/L or lower would have been even more common if it were not for the fact that many of the homes examined had very high initial radon levels. Based on this experience, the data suggest that available mitigation technologies are able to reduce radon levels in homes above 4 pCi/L down to 2 pCi/L, on average. Such reductions can be expected in houses with different foundation types, including basement, slab-on-grade, and crawlspace homes.

Similar evidence of the ability of mitigation techniques to reduce radon levels was compiled in ORP's Private Sector Radon Mitigator Survey, a nationwide survey of about 340 private sector radon mitigators in the fall of 1989. Exhibit 4-1 summarizes the final results from this survey for detached houses.[2]

EXHIBIT 4-1
PERCENT OF PRE- AND POST-MITIGATION RADON LEVELS CITED IN PRIVATE SECTOR RADON MITIGATOR SURVEY
(U.S. EPA/Radon Division 1990a)

PRE-MITIGATION		POST-MITIGATION	
pCi/L	Percent Homes	pCi/L	Percent Homes
< 4	1%	< 1	41%
4 - 10	23%	1 - 2	28%
10 - 20	37%	2 - 3	18%
20 - 50	29%	3 - 4	10%
> 50	10%	> 4	3%

[2]The question asked of private mitigators was "What were the radon concentrations in the last building you mitigated of each of the following types: detached house, school, workplace?" The answers for detached houses were used to develop Exhibit 4-1.

The exhibit shows mitigators reporting that they have reduced radon levels to below 4 pCi/L in 97 percent of the cases reported. Mitigators have also reduced radon levels to 2 pCi/L, or less in 69 percent of the cases reported. Examination of the pre-mitigation radon levels in these homes suggests that the private mitigator survey is reporting results from a set of homes that have a higher proportion of units with elevated radon levels (above 10 pCi/L) than will exist nationally in the homes that remain above the action level (i.e., 4 pCi/L) and therefore need to be mitigated. Also, although mitigators were trying to get radon concentrations below EPA's action level, they were not necessarily trying to reduce radon levels to 2 pCi/L or less. Therefore, EPA expects even better results for the owners of the remaining homes that follow EPA's advice and have mitigators try to reduce radon levels as much as possible. EPA believes that it is reasonable to assume that mitigators should be able to get all homes above 4 pCi/L down to an overall average of 2 pCi/L (with some homes still above 4 pCi/L and many homes attaining levels below 2 pCi/L).[3]

Based on the findings of ORD's own research and the mitigator survey, the Agency concludes that radon mitigation in the vast majority of cases (more than 95% of the time) should lead to radon reductions that leave homes with an annual average level of less than 4 pCi/L. It also will be very common for homes to have post-mitigation levels of 2 pCi/L or less. Results from the mitigator survey indicate that 2 pCi/L can be achieved approximately 70 percent of the time, while the ORD research suggests this estimate may be even higher. ORD's research shows that, on average, homes above 4 pCi/L can be mitigated to 2 pCi/L. In the fraction of homes where this lower level can be reached, reductions to 2 pCi/L or less should occur without inordinate effort, as the private sector survey results are based on mitigation experience when radon mitigators were trying to bring homes down to an action level of 4 pCi/L. Therefore, homeowners should expect that mitigators can provide radon reductions at, or well below EPA's action level.

Lowering high radon levels often requires considerable technical knowledge and special skills. As a result, the revised *Citizen's Guide* recommends that homeowners use mitigation contractors who have passed the EPA Radon Contractor Proficiency (RCP) Program tests. Research indicates that remedial actions taken by homeowners themselves are generally less effective than mitigation by contractors, and often are not followed by retesting to verify their effectiveness (Doyle et al. 1990). Nevertheless, EPA recognizes that some homeowners may want to mitigate their homes themselves and has produced several publications to help guide "do-it-yourself" efforts. The revised *Citizen's Guide* also recommends that homeowners contact their State radon office for further information on do-it-yourself mitigation.

Cost

The cost of a mitigation technique depends primarily on two factors: the cost of installing the system and the cost of operating and maintaining the system. Installation costs vary depending on foundation type, size and structure of the home, and initial radon level. Operating costs vary depending primarily on whether there is an energy penalty resulting from the loss of conditioned air, the price of electricity to operate active systems, and various maintenance requirements. Therefore, operating costs are usually presented as a range.

[3] Additionally, when EPA considered setting the action levels as low as 2 pCi/L, the Agency recognized that over 60 percent of the homes that should be fixed when the action level was 2 pCi/L would be between 2 and 4 pCi/L. These homes would only need to achieve a 50 percent reduction, or less, to meet that action level. Often they would probably have reductions well below 2 pCi/L.

From EPA's Private Sector Radon Mitigator Survey, the mean of the average reported mitigation cost in 1989 was approximately $1,200 and the median was $1,100.[4] Cited average costs ranged from $50 to $5,000, but about 85 percent of the average costs reported were between $500 and $2,000. More recent EPA research (U.S. EPA 1991b) suggests that it may be more appropriate to consider $2,500 as the upper end of the range.

Exhibit 4-2 presents estimated ranges of installation and operating costs for various dominant mitigation techniques. This exhibit is a result of the collaborative effort of EPA's ORP and ORD, and is based largely on the results of past research. The costs shown are typical ranges for most homes, although the actual cost of installing and operating a mitigation system in a home may be more or less than the range indicated in the exhibit. For active sub-slab depressurization, the exhibit's figures are further backed by a detailed study of the installation and operating costs of soil depressurization techniques (U.S. EPA 1991). More details on different aspects of radon mitigation are provided in Appendix F, which supports Chapter 5.

As shown in Exhibits 4-2, typical installation costs for radon mitigation techniques range from $100 to $3,000. However, depending on the type of house, installation costs can be above the upper end of this range. In addition to installation costs, most mitigation systems (except sealing) will have annual operating costs ranging from around $70 to $700. The effectiveness of each system will depend on the characteristics of the house to be mitigated.

[4]The question asked in the survey was "What are the average charges for the mitigation techniques you use in detached houses?" EPA then calculated the mean and median of the responses.

EXHIBIT 4-2
ESTIMATED INSTALLATION AND OPERATING COSTS FOR VARIOUS RADON MITIGATION TECHNIQUES (1991 $)[a]

Technique	Typical Radon Reductions	Typical Range of Installation Costs (Contractor)	Typical Operating Cost Range for Fan Electricity and Heated/Cooled Air Loss (Annual)	Comments
Sub-slab Suction (Sub-slab Depressurization)	80 - 99%	$800 - 2,500	$75 - 175	Works best if air can move easily in the material under the floor slab.
Drain-tile Suction	90 - 99%	$800 - 1,700	$75 - 175	Works best if drain-tiles form complete loop around the house.
Block-wall Suction	50 - 99%	$1,500 - 3,000	$150 - 300	Only in houses with hollow block-walls; requires sealing job of major openings.
Sump Hole Suction	90 - 99%	$800 - 2,500	$100 - 225	Works best if air can move easily to sump under slab, or if drain-tiles form complete loop.
Sub-membrane Depressurization in Crawlspace	80 - 99%	$1,000 - 2,500	$50 - 175	Less heat loss than natural ventilation in cold winter climates.
Natural Ventilation in a Crawlspace	0 - 50%	$200 - 500 if additional vents are installed; $0 if no additional vents	May be some energy penalties	Costs are variable
Sealing of Radon Entry Routes	0 - 50%	$100 - 2,000	None	Normally used in combination with other techniques. Requires proper materials and careful installation.
House (Basement) Pressurization	50 - 99%	$500 - 1,500	$150 - 500	Works best with tight basement that can be isolated from outdoors and upper floors.
Natural Ventilation	Variable	$200 - 500 if additional vents installed; $0 if no additional vents	$100 - 700	Significant heat and conditioned air loss; operating cost dependent upon utility rates and amount of ventilation.
Heat Recovery Ventilation	25 - 50% if used for full house; 25 - 75% if used for basement	$1,200 - 2,500	$75 - 500 for continuous operation	Limited use; works best in a tight house and when used for basement; less conditioned air loss than natural ventilation.

[a]The costs provided in this exhibit represent the range of typical costs for reducing radon levels in homes above 4 pCi/L down to radon levels below 4 pCi/L. In most cases homes are reduced to an average of about 2 pCi/L.

CHAPTER 5

COST-EFFECTIVENESS ANALYSIS

This chapter presents an analysis of the risk reductions and costs for the public if it followed the radon testing and mitigation advice that EPA offers in the revised *Citizen's Guide to Radon*. The first section of the chapter provides an initial overview of the analysis. This overview is followed by a section that describes the testing and mitigation options that the Agency considered in revising the *Guide* as well as the portion of housing stock and residential population that would be targeted by each. The next two sections analyze, in turn, the risk reductions and costs for the public that result from the options that EPA considered. The final section examines the cost-effectiveness of each alternative.

THE AIMS OF THIS CHAPTER

(1) To present an analysis of the risk reductions and costs of the testing and mitigation advice EPA recommends to the public.

(2) To enable an evaluation of the cost-effectiveness of the options considered.

Section I: Overview of the Analysis

This analysis estimates the annual risk reductions that would occur if the public followed the program prescribed in the revised *Citizen's Guide to Radon*, the annualized costs the public would incur, and the overall cost-effectiveness of the prescribed program as reflected by the cost per life saved. The analysis considers three options, with each consisting of the basic testing approach recommended in the revised *Guide* (Option C as defined in Chapter 3) coupled with different action levels for triggering the mitigation of a home. EPA recognized that the other testing options discussed in Chapter 3 also could be cost-effective in providing large risk reductions if the public followed the advice it had provided, but believed that these options had other weaknesses as outlined earlier that ruled out their selection. Following EPA's Science Advisory Board's concurrence on using Option C, the Agency focused its cost-effectiveness analysis on the action levels it should chose rather than on testing approaches.

The first option includes an action level of 4 pCi/L, which is the action level recommended in the revised *Citizen's Guide*. However, as stated in several places in the revised *Guide*, EPA recognizes that mitigation down to lower levels may be appropriate because radon levels less than 4 pCi/L still pose a health risk. Also, most homes today can be reduced to 2 pCi/L or below, and Congress has set a long-term goal that indoor radon levels be no more than outdoor levels, which are typically below 2 pCi/L. As a result, the cost-effectiveness analysis also considers two other options with lower action levels, 2 pCi/L and 3 pCi/L.[1]

[1]The Agency also examined action levels of 8 pCi/L and 20 pCi/L. These action levels are discussed in Appendix H.

The results for each option are compared to the figures that EPA's *Guidelines for Performing Regulatory Impact Analysis* (1983) suggest that the public is willing to pay to save "statistical lives." The figures presented in these guidelines represent the value the public places on reducing risks of death from all types of causes, given empirical evidence of the public's willingness to either pay to reduce small risks or receive payments for accepting those risks. The results also are compared to the cost-effectiveness of EPA's prior testing and mitigation advice in the original *Guide* and to the cost-effectiveness of other health and safety programs.

The analysis examines seven major components for each option: (1) the population of homes and people covered by the advice; (2) radon distributions in the affected homes; (3) the number of homes that should mitigate given the effectiveness of radon testing; (4) the risk reduction associated with lower radon exposure due to mitigation; (5) the unit costs associated with testing and mitigating homes; (6) the total costs of testing and mitigation for the group covered; and (7) the costs per life saved. The major steps in the analysis are summarized in Exhibit 5-1.

EXHIBIT 5-1
MAJOR STEPS IN THE COST-EFFECTIVENESS ANALYSIS

Step 1:	Determine the Universe Covered by the Testing and Mitigation Advice Option
Step 2:	Estimate the Radon Distribution in Affected Housing Units[a]
Step 3:	Calculate the Number of Housing Units that Should Mitigate Given the Estimated Accuracy of Radon Testing
Step 4:	Analyze the Annual Risk Reductions Resulting from Mitigation
Step 5:	Determine the Unit Costs of Testing and Mitigation
Step 6:	Develop the Total Annualized Costs of Testing and Mitigation
Step 7:	Calculate the Cost per Life Saved

[a] See Appendix D for a description of the affected housing units.

Throughout the analysis, a 100 percent public compliance rate with EPA's advice on testing and mitigation is assumed. There is no consideration of actions the public already has taken in response to the 1986 *Citizen's Guide* and independent State programs.

Section II: EPA's Approach to Reducing Residential Radon

This section provides a brief description of the residential radon problem, which, as opposed to radon problems in schools and other buildings, is the focus of the revised *Citizen's Guide*. It also describes EPA's advice to the public in the revised *Citizen's Guide* on how to reduce the risks from radon exposure and defines the residential population (and housing stock) that the *Guide* is trying to convince to take action to reduce radon exposures.

Residential Radon Problem

Radon can pose a significant health problem in any building where it can accumulate. Its greatest risk to the public appears to result from its accumulation in homes, where the public is estimated to spend about 75 percent of its time. Radon can emanate from certain building materials or be released at harmful levels in the home from water taken from ground-water sources, but it usually reaches elevated levels in homes due to soil gas entry through the home foundation. EPA's revised *Citizen's Guide* focuses on testing for radon in homes and taking actions to reduce radon entry or mitigate its effects. The Agency estimates that radon levels in the United States housing stock lead to 7,000 to 30,000 lung cancer deaths in the residential population annually. EPA's central estimate is that about 14,000 lung cancer deaths result each year from residential radon levels. The vast majority of the exposure occurs in single-family homes, mobile homes with permanent foundations, and the lower levels of apartment buildings and group quarters, such as college dormitories. Chapter 2 describes EPA's risk estimates for residential radon in more detail and Appendix B provides basic background information on radon.

Revised *Citizen's Guide's* Approach

In the revised *Citizen's Guide*, EPA advises homeowners initially to conduct a short-term test in the lowest lived-in level of their homes. If the results of the test are above the established action level of 4 pCi/L, homeowners are advised to obtain a follow-up measurement using either a short-term or long-term test. The higher the initial short-term result, the greater the certainty that the long-term average is also above the action level. With this in mind, and to keep homeowners from being unduly exposed to extremely elevated levels, EPA recommends that a short-term follow-up test be used if the results of the first short-term test are above 10 pCi/L. If homeowners conduct a long-term follow-up test, they are advised to mitigate their homes if the results of the long-term test are at or above the action level. If homeowners conduct a short-term follow-up test, they are advised to consider mitigating their homes if the average of the results of the two short-term tests is 4 pCi/L or more. The revised *Guide* also notes that homeowners can further reduce their lung cancer risk by mitigating homes that are below 4 pCi/L.

Homeowners are advised to mitigate their homes as quickly as possible. The mitigation technology that a person chooses to install will depend on the type of housing unit, the number of floors in the unit, the foundation type, and its interior space use. Most people will probably use one of the major technologies such as active subslab depressurization, sealing/plugging, natural ventilation, and heat recovery ventilation. Chapter 4 contains a more detailed discussion of these technologies.

In the revised *Citizen's Guide to Radon*, EPA establishes an action level of 4 pCi/L. The action level, i.e., the radon level at which homeowners are advised to mitigate, effectively determines the number of homes that should mitigate. Lowering the action level increases the number of homes that should mitigate and thus increases the scope of the radon program as well as the national risk reductions and total costs associated with the program. As described in the overview, this analysis also examines two alternative action levels, 2 pCi/L and 3 pCi/L.

Coverage of Revised *Citizen's Guide*

Exhibit 5-2 presents the total number of housing units and the total population that would be affected by EPA's advice to test and fix homes for radon. In the revised *Citizen's Guide to Radon*, all single-family homes, apartment units that are below the third floor, mobile homes with permanent foundations, and units in group quarters (e.g., college dormitories, military barracks) that are below the third floor should test for radon. EPA took the numbers of single-family homes, apartments in multi-unit structures, and mobile homes that were used in the analysis of the radon testing population from initial releases of 1990 Census data provided by the Bureau of the Census.[2] EPA's radon testing policy would lead to the testing of about 83 million housing units that are occupied by about 215 million people. This represents almost 83 percent of the housing stock that is regularly used and about 87 percent of the residential population. Appendix D provides greater explanation of how the estimates in Exhibit 5-2 were derived.

Section III: Risk Reduction

The risk analysis calculated the risk reductions resulting from the mitigation of homes with elevated radon. EPA estimated the annual risk reduction that occurred from each option examined in terms of annual lung cancer deaths averted, i.e., annual lives saved. The Agency also calculated annual deaths from lung cancer that are not prevented because of false negative test results (i.e., negative test results for a home that actually has a radon level above 4 pCi/L). The number of homes mitigated, and thus the risk reductions that occur, depends on (1) the action level established by EPA in each option, (2) the distribution of radon in housing units that test, and (3) the effectiveness of the radon tests. It also critically depends on the effectiveness of mitigation techniques that reduce radon levels.

This section initially describes the analyses conducted to determine how effective EPA's testing approach would be in classifying homes correctly using testing Option C (defined in Chapter 3) at various action levels. This description is followed by a summary of the assumptions made about the effectiveness of mitigation actions in lowering radon levels. The section closes with a explanation of how EPA calculated risk reductions based on this information and provides the results.

Results of Radon Testing

According to EPA's testing advice, an initial short-term screening test should be conducted in the lowest lived-in level of a home. If the results of the initial test are above the action level established by EPA, either a short-term or long-term follow-up test should be conducted. To determine the frequencies with which each type of test would actually be used by the public to make the follow-up measurement, EPA conducted a study of the public's willingness to use long-term tests. The study concluded that only 9 percent of the public were willing to use long-term tests (Johnson 1990). Accordingly, this analysis assumes that 9 percent of all follow-up tests will be long-term tests and 91 percent will be short-term tests.

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[2]EPA used 1990 Census data released in June 1991 (from Summary Tape File 1A) that provided official Census disaggregated estimates of population characteristics and types of structures as of April 1990. The Census Bureau's residential population estimate at that time was 248.7 million people, which by October 1990 was estimated to have grown to 250 million people (the basis of Chapter 2's risk calculation). As of May 1992, the residential population is 255 million people (Bureau of the Census, 1992b).

EXHIBIT 5-2
1990 HOUSING UNITS AND RESIDENTIAL POPULATION COVERED BY EPA'S RADON TESTING POLICY
(in 1000s)

HOUSING UNITS	Total Units[a]	Total Units Intended for Regular Use[b]	Units that Should Test for Radon[c]	Coverage Criteria from Radon Testing Policy	Total Residential Population[d]	Radon Testing Population
Single-Family	65,762	62,461	62,461	All single family homes	174,891	174,891
Multi-Units	27,981	26,569	17,801	All units below 3rd floor	52,596	35,239
Mobile Homes	8,521	8,067	403	All units on permanent foundation	14,526	726
Group Quarters	3,383	3,383	2,267	All units below 3rd floor	6,698	4,487
TOTAL	105,647	100,480	82,932		248,710	215,344
Percent of All Units	100%	95%	78%			
Percent of Units Intended for Regular Use/ Percent Population		100%	83%		100%	87%

Note: Totals may not sum due to rounding.

[a] All estimates, except group quarters are from 1990 Census data. Number of group quarters estimate based on assumed occupancy of about two persons per unit and 1990 Census estimate of persons living in group quarters. Mobile homes also include relatively small numbers of trailers and "other" units that could not easily be removed from the estimate.
[b] All estimates derived from available 1990 Census data on numbers of total units and excluding units with vacancies that were due to seasonal, occasional, and "other" (undefined) uses. All these cases were assumed to be situations in which owners would not test their units since they were not intended to have regular usage. Vacant units for rent or sale for year-round use were covered and used in estimating the units that should test in the next column.
[c] Estimates of units that should test based on EPA testing policy (i.e., that anyone living in detached houses, including mobile homes with permanent foundations should test for radon (U.S. EPA/Office of Public Affairs 1988)). Applicable criteria appear in the next column.
[d] Occupancy rates were derived from published and unpublished Census data and assumptions about similarities in unit usage between structures that did and did not have data available for direct computations of occupancy rates. These rates were used to estimate the population residing in each type of unit.

EPA generated distribution tables for each action level based on testing devices used, number of home floors, and testing conditions.[3] (For a more detailed explanation of radon tests, please refer to Chapter 3.) Based on 1990 Census data and data from the *Characteristics of New Housing: 1990 Current Construction Reports* (Bureau of the Census 1991), the weights for single floor vs. multi-floor homes were determined to be 46 and 54 percent, respectively. The same approach that was used to derive the mitigation classification rates in Chapter 3 was also used here, except the focus was on options that had different action levels combined with the same Option C testing approach.

Exhibit 5-3 presents the weighted testing distribution for the action level of 4 pCi/L. The distribution is based on the <u>actual</u> annual average radon level in the home and the radon level

EXHIBIT 5-3
JOINT PROBABILITY DISTRIBUTION OF RADON TESTING RESULTS FOR AN ACTION LEVEL OF 4 pCi/L

Intervals of Actual Annual Average Radon Level (pCi/L)	Radon Level According to Test Results (pCi/L)									TOTAL[a]
	0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	
	TRUE NEGATIVES - 90 %				**FALSE POSITIVES - 3.6%**					
0-1	0.4646	0.0931	0.0386	0.0179	0.0035	0.0012	0.0005	0.0004	0.0000	0.6198
1-2	0.0726	0.0661	0.0375	0.0216	0.0060	0.0021	0.0008	0.0006	0.0000	0.2074
2-3	0.0094	0.0195	0.0191	0.0134	0.0058	0.0025	0.0010	0.0009	0.0001	0.0717
3-4	0.0020	0.0069	0.0086	0.0099	0.0052	0.0029	0.0013	0.0013	0.0001	0.0382
	FALSE NEGATIVES - 2.2%				**TRUE POSITIVES - 4.0%**					
4-6	0.0007	0.0036	0.0061	0.0075	0.0084	0.0048	0.0027	0.0031	0.0003	0.0373
6-8	0.0001	0.0005	0.0011	0.0016	0.0018	0.0029	0.0015	0.0023	0.0004	0.0121
8-10	0.0000	0.0001	0.0003	0.0005	0.0007	0.0010	0.0015	0.0020	0.0005	0.0067
10-20	0.0000	0.0000	0.0001	0.0002	0.0003	0.0006	0.0007	0.0031	0.0012	0.0062
>20	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000	0.0001	0.0004	0.0006
TOTAL[a]	0.5493	0.1899	0.1114	0.0726	0.0318	0.0181	0.0101	0.0139	0.0030	1.0000

[a] Totals may not sum due to rounding. Very small values rounded to zero in the exhibit.

[3] The Agency generated distribution tables using information in a report entitled 'An Evaluation of the Performance of Alternative Short-Term Radon Testing Procedures in Homes with Pending Real Estate Transactions'. Chmelynski, H. 1992.

indicated by the test results.[4] From this exhibit, it is apparent that close to 8 percent of the housing stock should mitigate their homes based on their test results (i.e., 3.6 percent with false positive results plus 4.0 percent with true positive results). About 53 percent of these houses have actual true positive results; 47 percent of the homes actually have false positive results. Very often homes with false positive results also will receive risk reductions when mitigated and, therefore, substantial benefits are still derived from fixing homes below the action level. (This is discussed further in the next section on mitigation effectiveness.) The Agency also prepared unique distributions for the other action levels. From these distributions, EPA calculated the fractions of the population that obtain false positive, true positive, false negative, and true negative test results.

Exhibit 5-4 presents estimates of the homes testing, retesting, and mitigating at various action levels, as well as the population in each set of these homes. Of the 83 million homes that should initially test, about 10 million homes (12 percent) would be expected to retest using EPA's selected action level of 4 pCi/L. About 6 million of these homes will have results above the action level and should mitigate (8 percent of homes initially testing). Close to 17 million people live in the homes that should mitigate. This is about 8 percent of the occupants in homes that should test and close to 7 percent of the residential population of the U.S.

EXHIBIT 5-4
ESTIMATES OF HOMES TESTING AND MITIGATING
AT SELECTED ACTION LEVELS AND RESIDENTS IN THOSE HOMES (in 1000s)

HOUSING UNITS	Initial Testing	Retesting	Mitigating
Action Level (pCi/L)			
2	82,900	21,900	16,000
3	82,900	14,400	9,700
4	82,900	10,000	6,400
RESIDENTS	**Initial Testing**	**Retesting**	**Mitigating**
Action Level (pCi/L)			
2	215,300	57,000	41,600
3	215,300	37,300	25,200
4	215,300	26,000	16,500

[4]This distribution reflects the results expected to be obtained using the testing advice in the 1992 Citizen's Guide. Therefore, it includes homes that have tested once and received a result below the action level and homes that have tested twice. Homes that retest with a short-term test average the results of the two tests. Homes that retest with a long-term test use only the results of the long-term test since long-term tests are assumed to always result in correct classification of homes.

Effectiveness of Mitigation

For the purposes of estimating the benefits (risk reduction) of various policy decisions, this analysis assumes that homes with annual average radon levels above 2 pCi/L that mitigate will reduce radon levels to a post-mitigation annual lived-in area level of 2 pCi/L.[5] This is a level of radon mitigation that EPA believes most homes can achieve on average, as discussed in Chapter 4.[6] Homes with average radon levels below 2 pCi/L that mitigate (due to false positive test results) are likely to receive some reduction in radon levels, but the amount cannot be quantified. Therefore, the Agency conservatively assumed no reduction in risks. The Agency estimated radon exposure for the testing population using results from EPA's National Residential Radon Survey (U.S. EPA/Office of Radiation Programs 1991a). EPA also derived the mean radon exposure for each interval in the testing population distribution using data from the Survey. The exposure reduction achieved in each interval from mitigation was then calculated using Equation 1.

$$\begin{matrix}\textit{Radon Exposure}\\ \textit{Reduction in}\\ \textit{pCi/L Interval}\end{matrix} = \begin{matrix}\textit{Mean Radon}\\ \textit{Exposure in Interval}\end{matrix} - \begin{matrix}\textit{Post}\\ \textit{Mitigation Level}\end{matrix} \qquad (1)$$

Calculation of Risk Reductions

This analysis assumes that the reduction in risk associated with a reduction in radon exposure of 1 pCi/L was 43.2 lives saved annually per million people receiving the reduction. This risk factor was derived from EPA's risk assessment methodology that is explained in Chapter 2.[7] The central estimate of the radon risk factor is used in this analysis, rather than the upper bound risk estimate. The upper bound estimate would have increased all the risk estimates by about 2.5 times and improved later cost-effectiveness estimates by about 60 percent. Using the central risk factor, the number of radon deaths averted per million people was calculated for each radon concentration interval as shown in Equation 2.

$$\begin{matrix}\textit{Radon Deaths Averted}\\ \textit{in Interval}\\ \textit{Per Million People}\end{matrix} = \begin{matrix}\textit{Radon Exposure}\\ \textit{Reduction in pCi/L Interval}\\ \textit{(From (1))}\end{matrix} \times \begin{matrix}\textit{Risk Reduction}\\ \textit{Factor}\end{matrix} \qquad (2)$$

[5]Mitigation is assumed to lower homes to an average of 2 pCi/L, regardless of the action level.

[6]This assumption is based on review of the results from post-mitigation testing, which is often conducted in the lowest lived-in level in a home. Therefore, it could potentially lead to a conservative estimate of the average reduction of radon throughout the home (i.e., it could lead to an underestimate of the risk reduction). In Appendix G, the sensitivity of this assumption is examined.

[7]EPA converted its central risk factor (2.24 x 10^{-4} lifetime risk) into an annual risk reduction factor for homes based on a conversion of 0.193 working level months per pCi/L and assuming an average life expectancy of 74 years.

Annual lives saved (due to both true positive and false positive test results) and annual lives lost (due to false negative test results) were then calculated for each radon concentration interval using the weighted testing distribution and the testing population as shown in Equation 3.[8]

$$\begin{array}{c} \textit{Annual Lives} \\ \textit{Saved (Lost)} \\ \textit{in pCi/L Interval} \end{array} = \begin{array}{c} \textit{Testing} \\ \textit{Population} \\ \textit{(in Millions)} \end{array} \times \begin{array}{c} \textit{Fraction} \\ \textit{of Population} \\ \textit{in pCi/L Interval} \end{array} \times \begin{array}{c} \textit{Radon Deaths Averted} \\ \textit{in Interval} \\ \textit{Per Million People} \\ \textit{(From (2))} \end{array} \quad (3)$$

Exhibit 5-5 presents the results of the risk analysis for all three action levels and provides an estimation of the percentage of risk that is above each action level.[9] The estimated 2,200 lives saved annually with an action level of 4 pCi/L is roughly 17 percent of the lives that could potentially be lost due to residential radon exposure.[10] The action levels of 2 pCi/L and 3 pCi/L would reduce the estimated number of lung cancer deaths by 23 percent and 19 percent, respectively. More detail on how the risk assessment was conducted is provided in Appendix E.

EXHIBIT 5-5
ANNUAL LIVES SAVED AND NOT SAVED
UNDER ALTERNATIVE ACTION LEVELS FOR MITIGATION

Action Level	Percentage of Pre-Mitigation Risk Above the Action Level	Annual Lives Saved (Due to True and False Positive Test Results)	Annual Lives Not Saved (Due to False Negatives)
2 pCi/L	55%	3,100	460
3 pCi/L	42%	2,600	750
4 pCi/L	32%	2,200	740

[8]Equation 3 is used to calculate both annual lives saved and annual lives lost. If the test results are true or false positives, lives are saved due to the reduction in radon exposure. If the results are false negatives, lives are lost due to no reduction in radon exposure.

[9]The results in Exhibit 5-5 are based on a central estimate of 13,430 radon-induced lung cancer deaths per year. This estimate was developed using the Census Bureau's residential population figure of 248.7 million people as of April 1990, which is lower than the figure of 250 million people as of October 1990 used in Chapter 2.

[10]The reduction is lower than the percentage of risk that exists in homes above 4 pCi/L because mitigation will, on average, lower homes to 2 pCi/L, and because false negative results prevent mitigation from fixing all homes above 4 pCi/L. (Additionally, fixing many homes with false positive results does lead to significant levels of risk reduction.)

Section IV: Costs

EPA estimated the costs of testing and the costs of mitigating of homes based on the most recent information available, and employed basic assumptions that are consistent with those used in the risk analysis. The Agency calculated all costs as total present value costs. Annualized costs were then derived to show the total costs of each alternative in annual constant dollars over the time period of the analysis. All testing and installation of mitigation systems were assumed to occur immediately. The time period used for the analysis to estimate operating and maintenance (O&M) expenses was 74 years. This is actually the average life expectancy of the U.S. population today and a time period representative of at least the average life of a home.[11] The Agency chose this time period as an analytic convention to make the analysis comparable to other EPA analyses that estimate cancer cases averted based on an assessment of the lifetime risks and then convert those estimates into annual rates of cancer incidence avoided. A three percent discount rate was used to calculate both the present value of the total costs and the annualized costs.[12]

Testing Costs

The analysis assumes that the average initial homeowner testing cost is $28. This is based on the assumption that half of the homeowners would use charcoal canister tests (approximately $20 per test) and half would use alpha track detectors (approximately $35 per test).[13] For homes that retest, the cost of a second short-term test was again assumed to be $28. Long-term tests were estimated to cost homeowners $100 per test.[14] Based on a study conducted by EPA on the public's willingness to use long-term tests (Johnson 1990), this analysis assumes that short-term tests are used in follow-up testing 91 percent of the time and long-term tests are used 9 percent of the time.[15] The number of homes that test initially is assumed to be all homes covered by the revised *Citizen's Guide* advice, i.e., approximately 83 million homes. The number of homes that retest was calculated from the distribution of radon in the affected population using Equation 4.

$$\begin{matrix}\textit{Number of Homes} \\ \textit{That Retest}\end{matrix} = \begin{matrix}\textit{Fraction of} \\ \textit{Homes That Initially Test} \\ \textit{Above Action Level}\end{matrix} \times \begin{matrix}\textit{Number of} \\ \textit{Affected} \\ \textit{Homes}\end{matrix} \qquad (4)$$

[11]Examination of housing data on demolition rates suggests that the average home life is longer than 74 years.

[12]The three percent discount rate is based on the opportunity cost of money to the homeowner recognizing that most of the expenses will result from yearly operating and maintenance expenses.

[13]Some homeowners will use more expensive devices, especially during real estate transactions. The assumption here is that the vast majority of homeowners will use charcoal canisters or alpha track detectors, since CRMs and electrets aren't "do-it-yourself" devices.

[14]This assumption is based on homeowners using an alpha track detector over a long period of time in a single location or multiple locations; using charcoal canisters over several seasons; or having tests conducted by a contractor. This estimate is meant to be an overall average cost of follow-up by the 9 percent of homes retesting with a long-term test, which is a very modest percentage of total costs. The testing costs cover actions of homeowners and do not include additional tests mitigators may perform on site while they fix a home.

[15]The sensitivity analysis presented in Appendix G looks at the annual lives saved and cost incurred if 100% of the public used long-term confirmatory tests.

Total initial testing and retesting costs were calculated as shown in Equation 5 and Equation 6.

$$\textit{Total Initial Testing Cost} = \textit{Number of Homes That Initially Test} \times \textit{Cost of Initial Test} \tag{5}$$

$$\textit{Total Retesting Cost} = \textit{Number of Homes That Retest} \times \left(\textit{Cost of Short-term Test} \times 91\% + \textit{Cost of Long-term Test} \times 9\% \right) \tag{6}$$

<u>**Mitigation Costs**</u>

The costs of radon mitigation were calculated using a model developed by EPA. The model includes both upfront costs of system installation and operating and maintenance costs. The model provides a weighted average cost of mitigation based on characteristics of the housing stock such as foundation type, radon levels in the homes to be mitigated, and mitigation methods employed. The model produces different weighted average costs for each action level. For an action level of 4 pCi/L, the model produces a lifetime (74 years) weighted average cost of about \$6,550 per home. About \$1,500 of this expense (close to 25 percent), is for upfront costs of correctly installing systems. Close to \$1,250 of this upfront cost is direct installation of the system and the remainder is for price estimation and post-mitigation activities conducted to ensure proper system operation. The largest portion of mitigation costs results from O&M costs and is roughly \$5,050, or about 75 percent of the total costs. The majority of this cost comes from active systems that use electricity and require homeowners to pay for increased ventilation in the home or additional space conditioning because of exhalation of air beneath the foundation due to depressurization efforts in subslab systems.

A similar upfront and O&M cost relationship exists in the costs for the lower action levels. The total weighted average costs in those cases are \$6,150 and \$6,350 for 2 pCi/L and 3 pCi/L, respectively. The lower costs are due to an increasingly larger percentage of homes at lower radon levels that enter into the calculation of the weighted average costs. In the majority of situations (especially in homes using ASD), the costs of reducing radon levels in homes above all three action levels down to 2 pCi/L is about the same. However, mitigations in the smaller set of homes that use sealing and ventilation technologies cost less at the lower action levels, because the mitigation systems need to provide lower percentage reductions in radon levels.

An explanation of how the model works is provided in Appendix F. The model was designed to estimate costs for mitigation contractors for single-family detached homes (nearly 69 percent of the homes covered). This analysis assumes that mitigation costs for other types of housing units are the same as for single-family detached homes. This may overestimate the costs for these other units, since it fails to recognize economies of scale and other factors in mitigating single-family attached homes, multi-units, group quarters, and mobile homes. Of the housing stock that should be tested for radon, these units represent 7 percent, 21 percent, 3 percent, and 0.5 percent, respectively. The analysis assumes that all radon mitigation system installations occur immediately and that the mitigation systems operate over the 74-year period of the analysis. In addition to routine annual operating costs, biennial testing and repair and replacement costs are considered in developing lifetime home costs (U.S. EPA/Radon Division, 1991). The cost model does not make a distinction between homes that are mitigated due to true positive test results and homes that are mitigated due to false positive results. The number of homes that are mitigated was calculated using Equation 7.

$$\begin{matrix}\textit{Number of Homes} \\ \textit{That Mitigate}\end{matrix} = \begin{matrix}\textit{Fraction of •} \\ \textit{Homes That Retest} \\ \textit{Above Action Level}\end{matrix} \times \begin{matrix}\textit{Number of} \\ \textit{Affected} \\ \textit{Homes}\end{matrix} \qquad (7)$$

Total mitigation costs were calculated using Equation 8.

$$\begin{matrix}\textit{Total} \\ \textit{Mitigation Costs}\end{matrix} = \begin{matrix}\textit{Number of} \\ \textit{Homes} \\ \textit{Mitigated}\end{matrix} \times \begin{matrix}\textit{Weighted Average} \\ \textit{Cost of} \\ \textit{Mitigation}\end{matrix} \qquad (8)$$

Total Costs

Exhibit 5-6 presents the results of the cost analysis for all three action levels for testing and mitigation. It shows that if the public fully followed EPA's advice with an action level of 4 pCi/L, the annualized cost is about $1.5 billion. About 6 percent of the cost would be for testing and about 94 percent cost would be for mitigation. About 87 percent of the testing costs is incurred in the initial round of testing.

EXHIBIT 5-6
NATIONAL COSTS OF ALTERNATIVE ACTION LEVELS FOR MITIGATION
(1000s of 1991 $)

Action Level	Initial Testing Cost	Retesting Cost	Mitigation Cost	Total Cost
Total Present Value of Costs				
2 pCi/L	2,322,000	757,000	98,171,000	101,250,000
3 pCi/L	2,322,000	495,000	61,726,000	64,542,000
4 pCi/L	2,322,000	346,000	41,839,000	44,507,000
Annualized Costs				
2 pCi/L	78,000	26,000	3,317,000	3,421,000
3 pCi/L	78,000	17,000	2,086,000	2,181,000
4 pCi/L	78,000	12,000	1,414,000	1,504,000

Section V: Cost-Effectiveness

EPA has used cost per life saved (cost per death averted) as a measure of effectiveness of the revised *Citizen's Guide* in reducing risks to the public. This section discusses how EPA calculated cost-effectiveness and provides the results for the alternative action levels considered in revising the *Citizen's Guide*. The section then evaluates these results in three different ways, by (1) relating the results to the public's willingness to pay to save a statistical life, (2) comparing the cost per life saved of radon reduction to the cost per life saved of other safety and health-related programs, and (3) comparing the cost-effectiveness of the approach taken in the revised *Citizen's Guide* to that of the 1986 (original) *Guide*. The section also analyzes the sensitivity of the results to major assumptions and parameters used in the analysis.

<u>Cost per Life Saved of Selected Approaches to the Revised *Guide*</u>

The average cost per life saved for each alternative was calculated using Equation 9.

$$\begin{matrix} \textit{Average Cost per} \\ \textit{Life Saved} \end{matrix} = \begin{matrix} \textit{Total} \\ \textit{Annualized Cost} \end{matrix} \div \begin{matrix} \textit{Annual Lives} \\ \textit{Saved} \end{matrix} \qquad (9)$$

Exhibit 5-7 presents the average cost per life saved for all three action levels. The exhibit also presents the incremental cost per life saved.[16] The incremental cost per life saved is the added cost per life saved of each increment of risk reduction that can be gained when a lower action level is selected.[17] The results of the cost per life saved calculations for each action level are best evaluated in relationship to the value the public places on risk reductions. It is also helpful to look at the cost-effectiveness of other health and safety programs. Simply comparing one action level against another in the analysis can lead to misleading conclusions. For example, although an action level of 2 pCi/L is the least cost-effective option analyzed here, an EPA radon program based on that level could be more cost-effective in protecting public health than other governmental actions to reduce other forms of risk. It could also be providing a risk reduction at a cost the public is generally willing to pay for safety and health measures. Therefore, it is important to evaluate the results in Exhibit 5-7 in the broader context of what government can do overall to reduce risks.

In addition, the incremental cost per life saved is a very valuable tool in evaluating the cost-effectiveness of an option because it clearly assesses the value of the additional cost incurred by expanding the program to gain additional risk reduction. In evaluating the cost-effectiveness of the options analyzed here, the incremental cost per life saved for each option is best compared to the value the public places on saving a statistical life.

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[16]The cost per life saved of additional action levels is provided in Appendix H.

[17]For example, the incremental cost per life saved of an action level of 3 pCi/L is the increased annual cost of moving from an action level of 4 pCi/L to 3 pCi/L ($677 million) divided by the increased annual risk reduction (about 400 lives saved) -- an <u>incremental</u> cost per life saved of $1.7 million. Alternatively, the <u>average</u> cost per life saved is the direct division of the annualized cost of a 3 pCi/L action level ($2.18 billion) by the annual lives saved (2,600 lives saved) -- an average cost per life saved of $0.8 million.

EXHIBIT 5-7
COST PER LIFE SAVED
UNDER ALTERNATIVE TESTING AND MITIGATION PROGRAMS

Action Level	Number of Lives Saved Annually	Annualized Cost (1000s of 1991$)	Average Cost per Life Saved (1000s of 1991$)	Incremental Cost per Life Saved (1000s of 1991$)
2 pCi/L	3,100	$3,421,000	$1,100	$2,400
3 pCi/L	2,600	2,181,000	800	1,700
4 pCi/L	2,200	1,504,000	700	700[a/]

[a/]Based on assumption that "no action" was the alternative EPA had to this action level.

Note: The central estimate of the radon risk factor is used in this analysis, rather than the upper bound risk estimate. The upper bound estimate would have increased all the risk estimates by about 2.5 times and reduced the cost-effectiveness estimates by about 60 percent.

Public Willingness to Pay to Reduce Risks

The cost per life saved indicates how cost-effective an alternative is in providing health benefits to the public. It should be compared to the public's willingness to pay to save a "statistical life," i.e., buy risk reductions. In the past, EPA's 1983 Regulatory Impact Analysis (RIA) Guidelines (U.S. EPA 1983) have indicated that the public appears to value a risk reduction that saves a life (in statistical terms) for between $600,000 to $9,900,000 in 1991 dollars. A more recent study (Fisher et al. 1989) suggests that the public places the value of a statistical life between $2,000,000 and $10,500,000 in 1991 dollars.[18]

All three alternatives analyzed here have average cost per life saved values well below the upper end for the value of a statistical life. In fact, when compared to the most recent study's results of the public's willingness to pay, all three options provide an average cost per life saved that is well below the lower end of the range.

Cost per Life Saved of Other Programs

The cost per life saved calculations for the alternatives examined in this analysis can be compared to the cost per life saved of other health-related programs in order to evaluate how cost-effective the radon program is in protecting public health relative to other governmental actions taken to reduce risk. Its value as an aid to decisionmaking is based on the premise that given the limited availability of resources, public agencies should try to use societal resources to purchase the greatest risk reductions possible for the public. The lower the cost per life saved of actions taken, the more risk reduction the government could provide for the resources society can spend for health and safety protection.

[18]This study was conducted by the same researchers who provided EPA with the range in the 1983 RIA Guidelines.

Exhibit 5-8 shows the cost-effectiveness of other health and safety programs that has been reported in various sources. The effectiveness measures are cost per life saved. Another similar measure that is often used to evaluate the cost-effectiveness of government programs is the cost per cancer case avoided, when there is less certainty about the percentage of the time the cancer cases involved will lead to fatalities.[19] EPA has calculated this latter measure for many of its regulatory decisions because Agency risk estimates routinely report cases avoided, rather than fatalities. An EPA-sponsored review of EPA decisions over the last decade shows that the Agency has often decided on regulatory controls that cost several million dollars per cancer case avoided (Travis et al. 1987).

EXHIBIT 5-8
COST-EFFECTIVENESS ESTIMATES FOR SELECTED HEALTH AND SAFETY PROGRAMS[a/]

Federal Program Area	Range of Cost Per Life Saved/Case Avoided (1000s of 1991$)	Source
Medical Screening/Care	63 - 510 per life saved	Cohen 1987
Highway Safety	100 - 3,300 per life saved	OMB 1991
Air Transportation Safety	100 - 1,600 per life saved	OMB 1991
Occupational Safety	100 - 74,000 per life saved	OMB 1991

[a/] These values represent the ranges for regulatory decisions and health programs that have been reported in the literature from the most authoritative sources found. They are not all inclusive and other sources may provide different ranges because of differences in analytic approaches and year in which studies were completed.

The studies that have provided these estimates could have significant differences in their methodologies for calculating risk reductions and costs. Therefore, care must be taken in comparing these results to each other and to the results for the radon program. For instance, most assessments of the benefits from EPA rules are based on estimates of risk reductions that use conservative risk factors analogous to the upper bound estimate for radon. If the upper bound estimate for the risk factor had been used in this analysis, the average cost per life saved would have been reduced by about 60 percent.

However, simple direct comparisons of the range of results in Exhibit 5-8 and the review of EPA's own past regulatory experience to the results for the revised *Citizen's Guide* provide a general sense of how cost-effective the radon program is. For example, the average cost per life saved for

[19] Often in occupational health, food safety, and environmental protection analyses, only the incidence of cancer cases rather than death is calculated, because the target site and specific cancer mortality rate are unknown. In general, one in every two cases of cancer results in a fatality.

an action level of 4 pCi/L is approximately $700,000. This falls well within the range of the cost per life saved of highway safety, air transportation safety, and occupational safety programs and regulation. It is slightly higher than the upper bound cost per life saved of medical screening and health care. The $700,000 cost per life saved also compares favorably to the expenditures that EPA has been willing to see incurred for risk reductions in the past.[20]

Comparisons to the *1986 Citizen's Guide*

EPA also examined the risk reductions, costs, and cost-effectiveness of the *1986 Citizen's Guide* and compared its findings to what could be expected under the revised *Citizen's Guide*. The original Guide called for a short-term test in the lowest "livable space" and general follow up with a long-term test. If the long-term test results were greater than the action level of 4 pCi/L, mitigation was recommended.

EPA believes that about six percent of homes that should be tested had actually been tested as of July 1991. EPA estimates that this has led to the saving of about 40 lives per year, with an annualized cost of close to $40 million. The cost per life saved, therefore, has been close to $900,000. The cost per life saved under the revised *Guide* is about 25 percent less. The main reason for this reduction is that following the new EPA testing advice reduces the chances of false positive results. Homes with false positive results often will install mitigation systems that could provide only relatively small, if any, risk reductions.

Sensitivity Analysis and Examination of Higher Action Levels

EPA examined the sensitivity of the results of its cost-effectiveness analysis to the assumptions it used for major parameters of the analysis. This analysis covered the action level that EPA selected for the revised *Citizen's Guide* (action level of 4 pCi/L.) The results of changing major parameters within the reasonable range of potential values that they could have still suggest that the radon program should be cost-effective. The results of the analysis are provided in Appendix G.

In keeping with the Congressional mandate in IRAA to consider lowering the action level for radon mitigation, the Agency has focused its attention on action levels of 4 pCi/L and lower. This also is in keeping with EPA's mission as a public health agency to reasonably minimize risks to the public. EPA believes it is reasonable to focus on these lower action levels since higher action levels would remove a relatively smaller portion of the public's residential radon risk. Furthermore, having an action level of at least 4 pCi/L was determined to be incrementally cost-effective early in the Agency's deliberations on the *Citizen's Guide*. Appendix H provides a summary of the analytic results on which EPA has based its conclusions.

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[20]The cost-effectiveness of environmental controls is in terms of cancer cases avoided and not in terms of lives saved. Any comparison of cost-effectiveness between the radon program and other environmental programs should take this difference into consideration.

CHAPTER 6

RISK COMMUNICATION

Section I: Introduction

Two conditions must be met for any successful voluntary public health program: (1) the public must believe that the behavior or substance in question poses a significant personal threat, and (2) the solution must be simple enough for the public to be willing to adopt. The goal of the *Citizen's Guide* is to stimulate public action by convincing people that radon poses a serious health threat that can be mitigated relatively easily. Public action entails testing for the presence of radon and mitigating if necessary. Because it is difficult to persuade people to take voluntary action, it is critical to find and use the best possible risk communication strategies. Risk communication research has helped EPA develop and refine the communication strategies used in the *Citizen's Guide* to ensure its success in educating the public and stimulating testing.

> THE AIMS OF THIS CHAPTER
>
> (1) To summarize major results of radon risk communication research.
>
> (2) To explain how these results, along with public comments on risk communication issues, have been integrated into the final revised *Citizen's Guide*.

Risks associated with high radon levels in homes began receiving national publicity about six years ago. Since then, a number of researchers have studied risk communication issues associated with radon (e.g., the reactions of homeowners to information about radon risks). Most of this growing body of research has been sponsored by EPA's Office of Policy, Planning and Evaluation (OPPE) in cooperation with universities, States, and others. These studies include risk communication testing of the Fall 1990 draft *Guide*.

When the original (1986) *Citizen's Guide* was prepared, a major concern was avoiding public panic. Accordingly, the tone of the *Guide* was factual and informative. This approach, however, resulted in public apathy, not panic. EPA estimates that about 5.4 million homes have been tested for radon as of July 1991.[1] This figure represents approximately six percent of the occupied homes in the United States that should test (including single family homes, mobile homes with permanent foundations, apartment units below the third floor, and units in group quarters such as college dormitories that are below the third floor). Furthermore, of the small percentage of those who test for radon, many do not mitigate. EPA estimates that between 140,000 and 170,000 homes in the U.S. have been mitigated by contractors as of July 1991.[2] These estimates represent less than three percent of the occupied homes believed to have radon levels above 4 pCi/L.

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[1]This estimate is based on an update of the National Residential Radon Survey (NRRS) June 1989 results. To update the NRRS estimate, EPA conducted a telephone survey of large primary measurement firms to obtain information on the number of tests performed since June 1989.

[2]This estimate is based on an update of the Radon Division's Private Sector Radon Mitigator Survey finding that almost 100,000 homes had been mitigated by contractors as of February 1990. The estimate as of July 1991 is based on information collected by EPA staff in informal discussions with mitigation firms, the Regional Radon Training Centers, and States.

Radon risk communication has provided useful insight into why public apathy exists and suggests directions for overcoming it. Six key findings, listed in Exhibit 6-1, have emerged from the available research that can be applied to the revision of the *Citizen's Guide.*

EXHIBIT 6-1
KEY RISK COMMUNICATION FINDINGS

1. Be prescriptive as well as informative.

2. Streamline guidelines on testing and mitigation to minimize barriers to public action.

3. Overcome public denial through the use of persuasive appeals such as concern for the family.

4. Provide an appropriate level of radon information, since too much or too little information may result in an undesired effect.

5. Personalize the radon threat with tangible, relevant comparisons to familiar risk.

6. Stress that radon problems can be corrected but do not overstate the ease of fixing them.

The remainder of this chapter discusses the evolution of the current *Citizen's Guide* from the standpoint of risk communication research and issues. The first section (Section II) reviews the major research results as they relate to the six key findings above. The next section (Section III) describes how this research was incorporated into the Fall 1990 draft *Guide.* The following section (Section IV) discusses the major public reactions to the Fall 1990 draft *Guide* and what changes were made to incorporate public comments into the final revised *Guide.* The concluding section (Section V) reiterates the main themes of the chapter.

Section II: Research Findings and Implications

This section provides a brief summary of the research that supports each of the key findings listed in Exhibit 6-1.

1. Be Prescriptive

The original *Citizen's Guide* was informative, emphasizing what was known about radon risks in order to help the citizen make an educated decision to reduce risks by testing and mitigating if necessary. Research has shown, however, that people are more responsive to information brochures that tell them exactly what they should do. A study of different radon information formats (Smith et al. 1987), for example, found that, of several approaches tested, the "command/quantitative approach" that combined clear directions for action with precise information seemed to perform best overall. "Messages" that tell people clearly what to do are the most satisfactory, and when required actions are not emphasized, the resulting uncertainty can delay or even prevent action. Smith et al. (1987) also found that "people dislike uncertainty and may use it as an excuse for disregarding a radon risk message that indicates a lack of complete understanding of a risk or disagreement among experts." Similarly, Fisher and Johnson (1990) concluded that brochures that told people what to do rather than asked them to make their own decisions resulted in greater satisfaction with the message received.

2. Streamline Guidelines

As outlined in Chapter 3, the original *Citizen's Guide* recommended that individuals perform (1) a short-term measurement in the lowest livable level of the home under "closed-house" conditions, and (2) confirmatory tests in the lived-in levels if the result of the short-term test was above 4 pCi/L. The recommended duration of the confirmatory test depended on the result of the short-term measurement and varied from less than one week to a full year. The original *Guide* then provided guidance on whether and how quickly citizens should take action to fix their homes based on the confirmatory test results. This "test-test-fix" strategy is comprised of several steps that provide multiple opportunities for homeowners to "drop out" of the process. Compared to actions required to protect against other household risks, such as fire and theft, radon-related protective action recommended in the original *Guide* was a "multi-faceted, technical, multi-person, multi-skilled, recursive process" (Doyle et al. 1990) that deters action to reduce exposure.

Research has shown that it is very difficult to convince people to take even the first step to test their home. Weinstein, Sandman, and Klotz (1987), for example, investigated a random sample of homeowners living in or near the Reading Prong area in New Jersey and their responses to the radon threat. The researchers found that few people in the sample had tested or planned to test their homes for radon, even though they lived in a known high risk area. In accordance with the recommendations in the original *Guide*, this hurdle essentially had to be overcome twice -- once for the initial test and again for the confirmatory test.

Research also indicates that people are especially unwilling to take a second measurement if it means having to wait for long-term test results. For example, the Wirthlin Group's February 1990 telephone survey found that only 9 percent of the participants were willing to conduct a one-year radon test and that 27 percent were willing to wait no more than two days for test results (Johnson 1990). Fifty-five percent of respondents preferred a two-day test to longer-term tests. In a similar study, researchers found that 60 percent of all respondents preferred a two-day test to longer-term tests, 37 percent of all respondents stated that two days is the longest time that they would be willing to wait for test results, and only 20 percent were willing to wait for one year (Bruskin Associates 1991).

Other data confirm that long-term follow-up testing protocols are ignored by the public. Radon industry representatives maintain that despite EPA's recommendations, people are not conducting long-term tests. According to a senior representative from the American Association of Radon Scientists and Technologists, less than 5 percent of all testing devices sold are used for long-term testing (Walker 1991). People who have the initiative to begin a long-term test often misplace the measuring device, lose the mailer, or forget about the test altogether over the course of the year.

The experiences of State radon programs also demonstrate the public's lack of interest in long-term testing. For example, the Pennsylvania Department of Environmental Resources actively advertises free long-term follow-up tests for all citizens with screening test results of 20 pCi/L or higher, yet virtually no one requests the free tests (Walker 1991).

In addition, informal telephone surveys of measurement companies have found that most long-term testing devices are purchased for government studies, not by private citizens. Due to the lack of demand for long-term testing devices, only a small percentage of testing companies offer them for sale. As of July 9, 1991, only 120 testing firms out of 711 applying to the latest round of the Radon Measurement Proficiency Program market devices that can be used for long-term testing, and many of the devices sold by these firms may also be used for short-term testing (Walker 1991).

Finally, there is conflicting evidence about whether people who have tested for radon are inclined to fix their home. Some studies indicate that some people may drop out of the process before taking this final step. Doyle et al. (1989), for example, analyzed the effectiveness of the mass media radon information and testing campaign conducted in Washington, D.C. in 1988. A grocery store chain that teamed up with a Washington, D.C. TV station in its "Radon Watch" campaign could not keep monitors in stock at a bargain price of $4.75. However, only about half of the buyers actually performed the test and only a fraction with elevated levels did any credible mitigation. Researchers estimated that, of those people in the target population that tested their homes and needed to mitigate, only about 8 percent actually mitigated.

Other studies, however, provide evidence that once people have tested their home, they are likely to mitigate if necessary. Weinstein et al. (1988), for example, surveyed 123 New Jersey homeowners who discovered at least two years in advance of the study that they had elevated radon levels in their home. Of those whose initial first-floor readings were 20 pCi/L or more, 93 percent made home modifications. Seventy-one percent of those with readings between 8 and 20 pCi/L and 62 percent of those with readings between 4 and 8 pCi/L also modified their homes. Additionally, a June 1990 survey of 300 people who had tested their homes found that, of the 135 who had radon levels above 4 pCi/L, 24 percent had acted to reduce radon levels and the remaining 76 percent indicated that they planned to act, either by mitigating or by doing a follow-up test (Wagner and Dickson 1991).

3. Use Persuasive Appeals to Overcome Denial

Like many other public health and safety issues (e.g., seat belts) the radon issue suffers from people's belief that "it can't happen to me." There is a significant amount of research supporting the principle that people tend to deny that radon could be a problem in their home or their community (e.g., Desvousges and Kollander 1986, Smith et al. 1987, Weinstein et al. 1987, Weinstein et al. 1989, Leferman Associates, Inc. 1989, and TBWA 1990). Smith et al. (1987) evaluated how homeowners in New York State responded to different informational brochures. The researchers found that participants believed that their personal radon risks were lower than general population risks of radon. Weinstein et al. (1989) evaluated the responses to two public opinion surveys conducted in 1988 and 1989. Respondents to these surveys claimed that radon was less likely to be a problem in their own homes than in a home selected at random in their community. Even in high risk areas, residents tended to see a radon problem in their homes as fairly unlikely.

Research also shows that people use information about radon to rationalize why they do not have a problem and to create excuses for inaction. For example, when told that radon is caused by uranium in soil, people hypothesize that their soil does not contain uranium. Told that a certain percentage of homes are likely to have a radon problem, they assume their home does not. If people learn that radon problems are related to housing characteristics, such as age or construction type, they assume that the characteristics of theirs is such that it is not likely to be affected. Finally, if they learn that radon problems may be more prevalent in certain geographical areas, they hypothesize that their area is not likely to have a problem.

Research has found that this tendency to deny that high radon levels can affect "me" can best be overcome by using a "persuasive appeal" that affects people on an emotional level and provides a powerful risk message. TBWA focus groups found that focusing on concern for family members was a highly effective persuasive approach.

In communications testing of the 1990 draft *Citizen's Guide*, Leferman Associates, Inc. (1990) assessed consumer response to two different appeals used in radon information brochures: "protect your family" and "dangerous intruder." Both of these themes were developed from TBWA and other research suggesting the need to emphasize family protection, perhaps through an unsettling approach, in order to stimulate action quickly before barriers are built in people's minds. In this communications testing, both approaches were successful in conveying the risks of radon as the brochures were found to be clear, easy to understand, and believable. Overall, more than half of the participants claimed that they would be likely to test their home after studying either brochure. Positive responses were higher, however, among those exposed to the "dangerous intruder" brochure than among those evaluating the "protect your family" brochure. The "dangerous intruder" brochure also evoked a higher level of immediate action. The researchers believed that the "danger" expressed in that theme evoked a stronger need to do something than the, perhaps, more passive suggestion to "protect."

4. Provide an Appropriate Level of Radon Information

Several risk communication studies have evaluated the success of providing various levels of information in radon brochures. The most successful level of information is a difficult parameter to evaluate because other variables (e.g., the way in which the information is presented) complicate the issue. Also, "success" can be measured in terms of "success in promoting testing" or "success in communicating accurate information." While no study has identified the "perfect" level of information, the available risk communication studies suggest that too little information can cause confusion and unnecessary concern and too much information can be overwhelming or create excuses for inaction.

Smith et al. (1987), for example, examined alternative radon information formats and found that homeowners who received a brief fact sheet rather than an informational brochure showed lower levels of learning and were sometimes unnecessarily concerned over low radon readings. They were also more likely to ask for more information and more willing to demand the services of a radon diagnostician (Fisher and Johnson 1990). In a later study by Smith et al. (1990), homeowners received one of six types of radon information material along with the result of their radon test. Although the study did not identify any one format as being more effective than the others, it again found that the brief fact sheet is ineffective. The fact sheet created concern about risk from radon when it was not warranted.

Other research, in contrast, has found that readers can use additional information to create excuses for inaction. Results of focus group testing conducted by TBWA (1989), for example, suggest

3. Use Persuasive Appeals to Overcome Denial

As summarized in Section II, two "persuasive appeals" designed to affect people on an emotional level, "protect your family" and "dangerous intruder," were found to be successful in conveying the risks of radon. EPA used both of these themes in the draft revised *Guide.* For example, the draft overview to the *Guide* included a number of references to "keeping your family safe." EPA focused heavily on the "dangerous intruder" theme throughout the Fall 1990 draft *Guide* because it was believed, based on the research summarized above, that a strong message was needed to overcome the public apathy that was met by the original *Citizen's Guide.*

4. Provide an Appropriate Level of Radon Information

The Fall 1990 draft *Guide* sought to strike an appropriate balance between too little information, which can result in confusion and unwarranted concern, and too much information, which can result in greater denial of radon risks and a greater tendency toward inaction.

5. Personalize the Radon Threat

To personalize radon risks, the Fall 1990 draft *Guide* included charts comparing radon risks to an annual number of chest x-rays. In addition, EPA attempted to tailor the *Guide* to people in different smoking categories by separating information on smokers and never smokers. The risks to former smokers were addressed in a footnote to the charts. The Fall 1990 draft *Guide* also provided quantitative information on risks and the number of lung cancer deaths attributable to radon to help people fully understand and appreciate the threat.

6. Stress that Radon Problems Can Be Corrected without Overstating the Ease of Fixing

EPA recognizes that while emphasizing the ease of corrective action may stimulate more people to test and/or mitigate, overstating the ease of such action could cost the program some credibility. Therefore, the 1990 draft *Guide* emphasized that radon problems in homes can be remedied without overstating the ease, or understating the cost of corrective action. For example, the *Guide* stated "the cost of repairing a radon problem in the home will vary from home to home. Most homes can be fixed for about the same cost as other common home repairs. A lot depends on the way the home is built." In a later section the *Guide* stated "the costs of making repairs to reduce radon will vary depending on how your home was built. And the extent of the radon problem. But in most cases, homes can be fixed for between $500 and $2,000."

Section IV: Reactions to the Fall 1990 Draft *Guide* and Changes Made in Response to Comments

Recognizing the significance of the proposed changes in the Fall 1990 draft *Guide*, EPA took direct steps to solicit input from the public, States, and other government agencies, and to carefully weigh all comments in preparing the final version. EPA announced the availability of the draft *Guide* in the Federal Register on September 24, 1990 and distributed 700-800 copies of the *Guide* and its TSD for external review and comment. Although the comment period officially closed at the end of October 1990, EPA continued to receive and accept comments well beyond that time and ultimately received 143 individual comment letters totalling over 700 pages. Virtually all sectors were represented by the comments, including 33 States, Congress, 14 Federal agencies, numerous medical and building associations, universities, the real estate industry, the radon testing and mitigation industry, and private citizens. A large fraction of the public comments centered on risk communication issues.

The final revised *Citizen's Guide* attempts to strike a balance between what the risk communication research indicates is necessary and additional information provided by public commenters. The final *Guide* addresses the concerns of the commenters, without sacrificing all of its persuasive strength. This section summarizes the major concerns expressed by commenters as they relate to each of the six key findings and describes how these concerns were incorporated into the final version of the *Guide*.[3]

1. Be Prescriptive

Several commenters found the testing recommendations in the draft *Guide* to be confusing, and many misinterpreted the guidelines to be advocating a "test-fix" strategy. In addition, there was significant confusion among the commenters caused by the draft *Guide's* suggestion to "consider fixing between 2 and 4 pCi/L." The commenters thought that this suggestion makes the final action level unclear, and that in effect, it sets a de facto action level of 2 pCi/L.

The recommendations on the inside cover of the draft *Guide* were designed to encourage public action by being brief, clear and easy to follow. The misinterpretation of the recommendations by some of the commenters indicated to the Agency that the format or brevity of the recommendations may have caused some confusion. Therefore, EPA revised its testing recommendations to be much more explicit. The recommendation to fix, printed in bold at the very beginning of the final *Guide*, is essentially the same as that in the draft version; only the wording has been revised slightly to make the *Guide* more clear. The testing recommendations in a later section of the *Guide* have been expanded and clarified, and a new section, "What your test results mean," has been added. EPA believes that the recommendations are now very clear, and that there is no room for misinterpretation.

With regard to the recommended action level, the 1992 *Guide* maintains 4 pCi/L as the level for triggering home mitigation (see Chapters 3 and 7 for EPA's specific reasons for choosing this action level). For example, the front inside cover the *Guide* states in bold: "Fix your home if your radon level is 4 picocuries per liter (pCi/L) or higher." The final *Guide* also explains that there is some health risk at levels below 4 pCi/L and that most homes today can be reduced to 2 pCi/L or below. However, based on its risk communication research, EPA believes that this explanation simply provides homeowners with added factual information to help them make informed decisions and will not result in significant confusion.

2. Streamline Guidelines

Several commenters objected to the draft *Guide's* testing strategy, which allowed homeowners to reach a mitigation decision based on short-term tests. The commenters believed that short-term

[3] EPA has prepared a separate response-to-comments document (Response to Public Comments on EPA's Draft "A Citizen's Guide to Radon") that summarizes these and other issues in more detail and provides direct Agency responses.

test results are not reliable and do not adequately approximate average annual radon levels, especially at lower radon concentrations. Several commenters stated that short-term measurements may differ from annual average results by a very large amount and could, therefore, lead the homeowner to make the wrong mitigation decision.

EPA recognizes that short-term tests are not as representative as long-term tests of average annual radon levels. However, as summarized above, research has shown that people are much more likely to conduct short-term tests than long-term tests. Also, it is usually infeasible to conduct long-term tests during a real estate transaction. Based on these practical limitations, the 1992 *Guide* discusses the relative accuracy of short-term vs. long-term tests, emphasizing the benefits of long-term testing, but allowing people to choose to conduct either a long- or a short-term follow-up test. This approach represents the best balance between obtaining radon measurements that closely approximate average year-round radon levels and ensuring that people do not drop out of the testing process. In this sense, EPA believes that the approach recommended in the final *Guide* will result in the greatest amount of radon risk reduction.

3. Use Persuasive Appeals to Overcome Denial

Many commenters addressed the degree to which EPA should attempt to stimulate the public to test and fix their homes. A few commenters believed that the persuasive approach was effective and/or appropriate, with at least one risk communication expert commenting that the tone should be even more persuasive. A majority of commenters, however, believed that the approach was too alarmist. Although the severity of this concern among the commenters varied, most believed that the approach could confuse or alarm the public. Commenters focused their critique on the use of the "intruder" strategy, stating that it was patronizing and misleading.

As discussed above, it was EPA's intent that the revised *Citizen's Guide* function more as a persuasive brochure than as an informational document, prompting people to test their homes rather than educating them fully on the technical aspects of radon. After weighing these comments, EPA revised the *Guide* to be more informational and less "startling," while still striving to stimulate public action. EPA also abandoned the "intruder" concept in favor of the "protect your family" theme. This theme is displayed on the cover of the final *Guide* as well as being referred to throughout.

4. Provide an Appropriate Level of Radon Information

The many commenters who addressed the level of technical information in the *Guide* agreed almost unanimously that more general and/or technical information should be added. The commenters differed, however, in their suggestions of what additional information should be included. Several commenters, for example, suggested that the *Guide* should provide more information on testing and mitigation systems and techniques. Other commenters recommended that the *Guide* list references for more information and indicate how these references can be obtained. Finally, several commenters advised that the assumptions, estimates, and uncertainties regarding risk should be better covered in the *Guide*. Many of the commenters addressing this issue represented State radon offices that were concerned that the release of the draft 1990 *Guide* would cause them to be inundated with telephone calls requesting more radon information.

In response to these concerns, EPA added a significant amount of information to the *Guide*. Because the Agency recognized that the inclusion of additional technical information risked sacrificing the effectiveness of the *Guide* as a persuasive tool, EPA sought to strike a balance in deciding how much information to add. The Agency believes that the final revised *Guide* contains a sufficient level of information while sacrificing as little as possible of the clarity, accessibility, and persuasive strength of the draft 1990 *Guide*.

EPA added several new sections to the *Guide*, such as how to use a test kit (including appropriate sampling locations and procedures), options for short-term and long-term testing, what to do with test and retest results, radon and home sales, radon and home renovations, and radon myths. Several sections also have been clarified and/or expanded, such as how radon enters a home (including radon in building materials), radon in water, how to test, how to mitigate, and the health risks associated with radon. The *Guide* does not include what would be an extremely long list of radon contractors but suggests that the reader call his or her State radon office for a list of EPA-approved contractors in the State. The *Guide* also includes a list of State radon office phone numbers and hotline numbers, and references additional EPA materials and indicates how they can be obtained. Also, in revising the *Guide*, an effort was made to address the assumptions and uncertainties associated with radon risk in such a way that informs homeowners but does not present a barrier to action. Based on recommendations made by EPA's SAB, the *Guide* includes several statements acknowledging uncertainty in the risk estimates. More specific information on uncertainty was not included because it was considered too technical. Instead, the *Guide* references this TSD, which contains a detailed and expanded discussion of uncertainty in Chapter 2.

5. Personalize the Radon Threat

The commenters agreed that the use of comparative risk charts was an effective way of conveying personal risks. Several commenters suggested that measures should be taken to decrease the chance of alienating nonsmokers or allowing smokers to discount their risk due to radon. In addition, many commenters suggested that the x-ray comparison is misleading and frightening and should be replaced by a more tangible analogy, such as the risks associated with accidents in homes or cars, or radiation doses from airplane flights.

The Agency's primary goal in designing a risk chart for the updated *Guide* was that it characterize risk as accurately as possible without alienating either smokers or nonsmokers or allowing them to deny their risk. To do this, the final version of the *Guide* presents two separate risk charts. The Agency believes that this approach allows both smokers and never smokers to accurately characterize their risk without alienating either group.

EPA had spent a considerable amount of effort prior to the release of the draft revised *Guide* identifying an effective and appropriate comparison to the health risks associated with radon. The Agency found in extensive communications tests that the comparison to x-rays did not frighten people. In fact, the risk communication research showed that people found the x-ray comparison clear and easy to understand. Nevertheless, in response to the concerns expressed by commenters, the Agency conducted additional communication studies on four possible comparisons: x-rays, illnesses, accidents (e.g., airplane crashes, home fires, etc.), and an approach that combined several comparisons. EPA found in this later research that none of the sets of risk comparisons was clearly better than another at producing appropriate responses to the risks from radon (Weinstein and Sandman 1991). In light of this finding and the concerns of commenters, EPA chose to compare radon risk to the risk of being killed in different types of accidents.

6. Stress that Radon Problems Can Be Corrected without Overstating the Ease of Fixing

Several commenters addressed the costs of mitigation as an issue in the draft revised *Guide*. Most of these commenters recommended that the *Guide* should be more explicit and provide specific remediation costs for different remediation methods, rather than just provide a range. A few commenters suggested the *Guide* provide examples of home repairs that are of comparable costs to radon mitigation, with one believing that the draft *Guide* exaggerated its claim that a home can be mitigated for the same cost as other home repairs.

The final *Guide* discusses remediation techniques and costs in the "How to Lower the Radon Level in Your Home" section. The Agency revised the range of $500-$2,000 estimated to fix a home to $500-$2,500, based on data collected by EPA's Office of Research and Development. These data augmented the Private Sector Radon Mitigator Survey that surveyed 340 radon mitigation firms on their radon reduction activities (U.S. EPA/Radon Division 1990a). The survey found that, within this range, approximately $1,200 (in 1989 $) was the average charge to mitigate a detached house. The *Guide* also provides examples of home repairs (i.e., painting or installing a new water heater) that are comparable to the costs of radon mitigation.

Section V: Summary and Conclusion

Risk communication research has made an extremely important contribution in the evolution of the *Citizen's Guide*. Over the past six years, a number of researchers have analyzed key issues relating to radon risk communication, and their findings were incorporated into the Fall 1990 draft *Guide*. Studies of the Fall 1990 version of the *Guide* showed the document to be extremely effective in terms of risk communication. When the draft *Guide* was distributed for review, commenters were able to identify areas in which they believed that other concerns should be considered and factored in with the risk communication findings. Commenters noted, for example, that while the "intruder" concept may be effective in stimulating testing, some people may find it offensive. Also, as State representatives pointed out, an effective level of information from a risk communication standpoint may result in State radon offices being inundated with calls from people asking for more information. Thus, the commenters helped EPA refine its use of the six key risk communication findings and develop a final version of the *Citizen's Guide* that both incorporates the main findings of risk communication and addresses the commenters' practical concerns.

CHAPTER 7

RATIONALE FOR 1992 *CITIZEN'S GUIDE*

This chapter summarizes the main conclusions from the previous chapters and briefly describes how those conclusions were used to develop the 1992 *Citizen's Guide*. It outlines the rationale underlying EPA's approach in the revised *Guide* for accomplishing two main goals: (1) communicating radon risks to the public; and (2) recommending a radon testing and mitigation strategy for homeowners. The chapter also discusses the interrelationship of these two goals and how EPA considered them together in shaping the 1992 *Guide*.

Section I: Risk Communication

Developing the final approach for risk communication in the revised *Citizen's Guide* required the consideration of two distinct issues: what to tell the public about radon risks and how to communicate that information. The rationale for the revised *Guide's* approach with respect to each of these issues is outlined in separate sections below.

What to Tell the Public about Radon Risks

As presented in Chapter 2, radon is one of the most extensively studied pollutants in the world. Radon risks have been well documented by numerous scientific and public health authorities, and are based on extensive studies of thousands of underground miners. This research provides strong scientific evidence that exposure to radon can cause lung cancer in humans. Further research since the original *Citizen's Guide* was published in 1986 has helped to improve EPA's projection of lung cancer risk to the general population due to radon exposures in the home. EPA now estimates that the number of lung cancer deaths per year in the U.S. due to residential radon exposure is approximately 14,000, with an uncertainty range of 7,000 to 30,000. Even using the lower end of this uncertainty range, this makes indoor radon the second leading cause of lung cancer in the U.S., after smoking. Continuing scientific evaluation of radon hazards serves primarily to refine EPA's estimate of the annual number of radon-induced lung cancer deaths, not to determine whether indoor radon exposure poses a serious public health problem.

In addition, in order to understand the risks of indoor radon, the public needs information on the relationship between radon and smoking risks and the radon risks to children. Key findings related to each of these issues are summarized below:

- Although the risk of lung cancer from radon exposure appears to be enhanced by smoking tobacco, a person does not have to be a smoker to be at risk from radon. EPA estimates that the radon risk for current smokers is close to 20 times the risk for never smokers and the risk to former smokers may be over 8 times greater than the risk for never smokers.

- Although there is information that indicates that children may be at greater risk than adults from some kinds of radiation exposure, there is no direct evidence that children are at greater risk than adults from radon.

Based on these conclusions, the revised *Guide* warns that an individual's lung cancer risk is especially high if he or she is a smoker, and provides risk charts that allow readers to identify their radon risk according to what smoking category they fall into. The revised *Guide* also clarifies that "children have been reported to have greater risk than adults of certain types of cancer from radiation, but there are currently no conclusive data on whether children are at greater risk than adults from radon."

How to Communicate Radon Risk Information

As discussed in Chapter 6, extensive risk communication research since the 1986 *Guide* was published has provided useful insight into why the public remains largely apathetic about indoor radon and suggestions for overcoming that apathy. In developing the revised *Guide*, EPA applied six key findings that have emerged from this research: (1) be prescriptive rather than simply informative; (2) streamline guidelines on testing and mitigation so they do not present barriers to public action; (3) overcome public denial through the use of "persuasive appeals" such as concern for the family; (4) provide an appropriate level of radon information, since too much or too little information can result in an undesired effect; (5) personalize the radon threat with tangible, relevant comparisons to familiar risks; and (6) stress that radon problems can be corrected but do not overstate the ease of fixing them.

In updating the *Citizen's Guide*, it was important for EPA to balance the results of this risk communication research with other practical considerations. For example, some of the risk communication research suggested that the revised *Guide* should provide only a minimal amount of technical information on radon, since readers can use additional information to create excuses for inaction. This finding, however, had to be balanced against concerns that State offices could be inundated with public requests for more information if the *Guide* did not provide enough detail to answer homeowners' questions. This balancing of the risk communication research with practical limitations led to the following design features with respect to the six key findings listed above:

- The 1992 *Citizen's Guide* was designed to be prescriptive. It provides brief, clear, and easy to follow directions on what to do (e.g., how to test and when to mitigate), rather than simply providing information and allowing readers to come to their own conclusions.

- EPA's Radon Program experience and risk communication research indicate that many people drop out of the testing and mitigation process before they fully comply with the 1986 *Guide's* recommendation to conduct a long-term follow-up measurement prior to reaching a mitigation decision. Based on this finding and EPA's detailed analysis of various testing options, the revised *Guide* attempts to streamline the testing guidelines by: (1) emphasizing the benefits of long-term testing; but also (2) allowing people to choose to conduct either a long- or short-term follow-up test.

- The 1992 *Guide* was designed to function more as a persuasive document than the 1986 version. It utilizes the "protect your family" theme found to be effective in risk communication testing, while avoiding other emotional appeals that might compromise the scientific credibility of the message.

- The revised *Guide* contains a sufficient level of technical information to educate homeowners, while sacrificing as little as possible in terms of clarity, accessibility, and incentive to test.

- EPA designed the revised *Guide* to help personalize the radon threat by providing tangible, relevant comparisons to familiar risks. Specifically, the revised *Guide* provides risk charts for smokers and never smokers that characterize each group's risk as accurately as possible without alienating them or allowing them to deny their risk. The *Guide* also compares radon risk to other risks, such as drunk driving, drowning, fires, airline and car crashes, and violent crimes.

- The revised *Guide* puts radon mitigation in proper perspective by describing remediation techniques, providing realistic estimates of remediation costs based on EPA's Private Sector Radon Mitigator Survey and research by EPA's Office of Research and Development, and providing examples of other home repairs that are of a comparable cost.

Section II: Testing and Mitigation Advice

The *Citizen's Guide* provides specific advice to homeowners on how to test for radon and when they should mitigate. EPA's rationale for this advice, drawing on the main conclusions from the previous chapters of this document, is summarized below.

How to Test

Chapter 3 identifies several major attributes that are used to define the recommended radon testing strategy, including: (1) a testing location, (2) ventilation conditions, and (3) a testing duration before reaching a mitigation decision. EPA's recommendations and rationale with respect to each of these attributes are summarized below, followed by an explanation of the basis for the overall testing strategy recommended in the revised *Citizen's Guide*.

1. Testing Location

EPA is revising the recommended test location for the initial short-term radon measurement from the "lowest livable" level, recommended in the 1986 *Guide*, to the "lowest lived-in" level. There is a difference between these two locations. The lowest livable level is the lowest area of a home that is used or has the potential to be used as a living space, whereas the lowest lived-in level is the lowest level of a house that is used regularly. In both the original and revised *Guide*, however, the recommended location for decision-making tests (i.e., tests on which to base a mitigation decision) is the lowest lived-in level.

Roughly 50 percent of the homes nationwide have basements; however, only about half of these homes, or almost 25 percent of the national total, have basements that are used as a lived-in level (U.S. EPA/Office of Radiation Programs 1991a). Accordingly, a recommendation that devices be placed in the lowest livable level would result in 50 percent of the devices being placed in the basement. A recommendation that devices be placed in the lowest lived-in level would result in only 25 percent of devices being placed in the basement, and the remaining 75 percent being placed on the first floor.

Keeping the short-term test in the lowest livable area as compared to moving it to the lowest lived-in area would reduce false negative results by a factor of 1.5; however, the number of false positive results would increase by a factor of 2. Measurements taken in the lowest lived-in area strike a more equitable balance between false positives (public money spent with more limited benefits in many cases) and false negatives (public health protection) than do lowest livable area measurements. Most importantly, lowest lived-in measurements are the most representative of human exposure to radon. Recent research by Harley et al. (1991) suggests that basement measurements overstate personal exposure by a factor of 3 to 5, while first floor measurements are, on average, only 30 percent greater than occupant exposure (for short-term measurements). Since the goal of radon measurement on which a mitigation decision relies is to assess occupant exposure (U.S. EPA 1991a), lived-in level measurements are better predictors of risk than livable level measurements. Based on this finding and the recommendation of the EPA Science Advisory Board (U.S. EPA 1992b), short- or long-term measurements that are to be used for the purpose of making mitigation decisions should be made in the lowest lived-in level because it more closely approximates the radon concentration to which inhabitants are exposed.

2. Ventilation Conditions

EPA examined whether tests should be made under "open-house" or "closed-house" conditions. In open-house conditions, tests are made with windows and other ventilating passageways either closed or open as they would be when a test is not being conducted. In closed-house conditions, windows and other ventilated passageways are closed to the extent possible. EPA stresses the value of closed-house conditions in the revised *Guide*, because these conditions provide more consistent radon measurements. However, the revised *Guide* makes allowances for practical limitations in maintaining these conditions. Specifically, the *Guide* recommends keeping windows and outside doors closed "as much as possible" during short-term testing.

3. Testing Duration

The 1986 *Citizen's Guide* recommended that homeowners perform (1) a short-term measurement, and (2) a confirmatory measurement if the result of the short-term test was above 4 pCi/L. The recommended duration of the confirmatory measurement depended on the result of the short-term test and varied from less than one week to a full year. Mitigation decisions were to be based on the results of the longer-term confirmatory measurement. Since the original *Guide* was published, however, EPA's Radon Program experience and risk communication research have indicated that many people drop out of this process before obtaining the long-term measurements needed to reach a mitigation decision. Short-term tests are much more appealing to the public because they are simpler to make and provide faster results. Therefore, it became apparent that a recommended testing procedure that does not rely solely on long-term testing may be more effective in actual risk reduction by removing an existing barrier to radon mitigation.

EPA examined the implications of relying on short-term tests as indicators of annual average radon levels in developing the 1992 *Guide*. From knowledge of the relationship between short-term measurements and annual averages gained from EPA/State Residential Radon Surveys, the Agency recognized that short-term measurements are imperfect indicators of annual averages, but believed they potentially could provide homeowners with a reasonably accurate basis for determining whether they need to mitigate. EPA thus constructed various testing options, including different test durations prior to reaching a mitigation decision, and analyzed the effectiveness of each option in terms of misclassification rates and public acceptability. Based on the results of this analysis, described in detail in Chapter 3 and summarized in the following section, EPA concluded that a testing strategy that allows homeowners to reach a mitigation decision based on short-term test results would maximize public health protection.

4. Rationale for Recommended Testing Strategy

EPA began its analysis of radon testing options by considering a strategy that requires a single short-term measurement followed by a confirmatory long-term test. This option, defined as Option A in Chapter 3, parallels the testing procedure recommended in the 1986 *Citizen's Guide*, except that both tests would be conducted on the lowest lived-in level. Under Option A, 98 percent of homes were correctly classified with regard to the need for mitigation. To determine how testing procedures relying on a short-term confirmatory test would compare to this option, EPA developed and analyzed five other options, defined as Options B through F in Chapter 3. EPA found that all of these options produced results yielding at least 94 percent correct classifications.

Given these results, the Agency determined that on the basis of misclassification, none of the options should be rejected. EPA decided instead that other factors, such as procedure simplicity, should be weighed with the trade-offs for increased levels of correct classification in selecting the testing option to recommend in the revised *Guide*. As noted above, current Radon Program experience indicates that few people are actually taking long-term follow-up measurements prior to reaching a mitigation decision, and that most people who do mitigate do so based on a single short-term test. This finding is supported by EPA's risk communication research that indicates that as little as 9 percent of the population is willing to conduct year-long tests. These conclusions suggest that a testing protocol that relies exclusively on long-term confirmatory tests is unlikely to be followed by most of the public. Therefore, although Options A and D have more desirable error rates than the other options, compliance with these options is likely to be low. Also, selection of either of these two options is apt to result in a situation in which decisions are most often made based on a single short-term measurement. By explicitly providing homeowners a process that calls for more than one short-term test, all of the other options considered would be more likely to result in people having better information to use in reaching a mitigation decision.

On the other hand, since long-term measurements are more desirable than short-term measurements and since there are some individuals who are willing to conduct long-term tests, the use of long-term tests should not be precluded from the testing protocol. As a result, Options B, E, and F, which do not include long-term tests, are also undesirable.

Option C offers an effective compromise between these different approaches. It promotes long-term testing by people who are willing to conduct long-term tests and recommends an effective short-term test as an alternative for people who are not willing to conduct a long-term test. Therefore, consistent with the advice from EPA's Science Advisory Board, the revised *Guide* recommends Option C because it should maximize the total risk reduction the public would gain through future testing and mitigation while minimizing error. The *Guide* also explains the trade-offs between short-term and long-term testing. It acknowledges that long-term testing is more representative of annual exposures, but enables the public to have a short-term measurement process that is sound.

When to Mitigate

EPA is recommending that the action level of 4 pCi/L established in the 1986 *Guide* be maintained for several reasons. First, lower action levels introduce more uncertainty in the measurement results. Measurement device error increases to approximately 50 percent at 2 pCi/L. This device error in conjunction with the larger fraction of homes (of total homes testing) that have radon levels around 2 pCi/L would result in a threefold increase in false negatives and a twofold increase in false positives over those expected at a 4 pCi/L action level. In addition, as outlined in detail in Chapter 4, the Office of Research and Development's (ORD's) research on mitigation effectiveness and the Office of Radiation Programs' Private Sector Radon Mitigator Survey suggest that elevated levels of radon can be reduced to 4 pCi/L more than 95 percent of the time. Results from the mitigator survey indicate that 2 pCi/L can be achieved about 70 percent of the time, while the ORD research suggests this estimate may be even higher (U.S. EPA/ORD 1989; U.S. EPA/Radon Division 1990a). Reducing the action level to 2 pCi/L, therefore, could result in perhaps as many as 30 percent of homes with elevated levels being unable to achieve the action level.

However, EPA recognizes that mitigation down to lower radon levels may be appropriate because levels below 4 pCi/L still pose a health risk. Furthermore, as mentioned above, mitigation technology available today permits most homes to be reduced to 2 pCi/L or below, and Congress has set a long-term goal that indoor radon levels be no more than outdoor levels, which are typically less than 2 pCi/L. As a result, EPA also closely examined the costs and benefits of selecting an action level of 2 pCi/L and 3 pCi/L.

The results of this cost-effectiveness analysis, detailed in Chapter 5, show that setting the action level at 4 pCi/L would result in a cost of roughly $700,000 per lung cancer death averted (i.e., per life saved). Lowering the action level would incrementally increase this cost to $1,700,000 per life saved if 3 pCi/L was used, and to $2,400,000 per life saved if 2 pCi/L was used instead of 3 pCi/L. All three of the action levels have cost per life saved values that are at the lower end of, or below, the values that the public is willing to pay to save a statistical life, according to EPA's 1983 *Regulatory Impact Analysis Guidelines*. Based on these findings, EPA believes any of the three action levels considered would provide cost-effective results.

At the selected action level, the Radon Program would be as or more cost-effective than many other government programs for personal safety and environmental protection. EPA also believes the recommended testing protocol combined with an action level of 4 pCi/L in the 1992 *Guide* will be more cost-effective than that recommended in the 1986 *Guide* — $700,000 per life saved now versus the $900,000 per life saved that EPA calculates from program experience based on the 1986 *Guide*.

EPA's decision to keep 4 pCi/L as the action level in the revised *Citizen's Guide* is supported by the cost-effectiveness analysis. However, the revised *Guide* also notes, although not with the same weight as the recommended action level, that homeowners can further reduce their lung cancer risk by mitigating homes that are below 4 pCi/L. As long as the revised *Guide* clearly establishes 4 pCi/L as the recommended action level to avoid confusion and the other problems mentioned above, EPA believes this discussion of reducing radon below the action level helps to fully inform the reader and is justified based on: (1) the health risk involved; (2) the effectiveness of available mitigation technology; (3) cost-effectiveness; and (4) Congressional intent.

Section III: Conclusions

In developing the 1992 *Citizen's Guide*, EPA had to balance the findings of its technical analyses on risk, testing accuracy, mitigation technology, and cost-effectiveness with the information it was collecting from its risk communication research. For example, EPA and the scientific community had amassed and analyzed in depth a considerable amount of information on the level and significance of indoor radon risk since the original *Guide* was published in 1986. EPA had to convey that risk in the updated *Guide* with a message that was strong enough to persuade homeowners to act, while being careful not to provide too much detail, which could sacrifice accessibility, or make it too startling, which might compromise scientific credibility and support. Furthermore, although EPA recognized the technical superiority of long-term versus short-term testing after extensive evaluation of the issue, it had to accept the compelling practical limitation that the public at large is more likely to use short-term testing. A lot of "good" testing, after all, will provide greater public health protection than a more limited amount of "perfect" testing. Finally, emphasis on an action level that is achievable by the vast majority of homes is better than recommending a lower action level that pushes the limit of technology. EPA's ultimate objective was to advance a technically well-supported 1992 *Citizen's Guide* that takes a pragmatic step in better communicating radon's risks to the public and promoting broader public action in response to the problem.

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APPENDIX A

RADON CONCENTRATIONS IN AMBIENT AIR

According to Section 301 of the Indoor Radon Abatement Act, "The national long-term goal of the United States with respect to radon levels in buildings is that the air within buildings in the United States should be as free of radon as the ambient air outside of buildings." To ensure that EPA program goals are consistent with the requirements of this legislation and to confirm reported outdoor ambient concentrations, EPA reviewed available literature on outdoor radon measurements and conducted the Ambient Radon Field Study (Hopper 1991).

Individually measured radon concentrations vary widely in the reviewed literature -- from 0.1 to 1.2 picocuries per liter (pCi/L) -- while the average concentrations range from about 0.1 to 0.8 pCi/L. In general, concentrations appear to be higher in the western States than in the eastern States. Articles describing measurements of ambient radon-222 concentrations obtained within one mile of a uranium mill tailings pile or phosphogypsum stack were not used in this analysis.

The first consideration when measuring the very low radon concentrations expected in outdoor air is the reliability of the measurement method at such low levels. The methods used to measure ambient radon concentrations include alpha track detectors, continuous radon monitors, continuous flow-through monitors (scintillation cells), pump/collapsible bag devices, and electret ion chambers (EICs). There are very few data establishing accuracy and precision of these methods at levels of 0.1 to 1 pCi/L. Because of this uncertainty, the data quality objectives for the EPA Ambient Radon Field Study were:

1. To determine if the majority of the ambient radon levels are < 0.6, < 1.0, or >1.0 pCi/L;

2. To maintain a precision in the measurement that will allow EPA to distinguish between an ambient air level of 0.2 pCi/L and 0.6 pCi/L; and

3. To assure that the 2-standard deviation error will be no greater than 0.25 pCi/L, at an ambient radon concentration of 0.25 pCi/L.

The study was designed to measure selected outdoor radon levels across the country. Since EPA's Office of Radiation Programs (ORP) has Environmental Radiation Ambient Monitoring System (ERAMS) stations in every State, these stations were selected as the sample sites for making quarterly outdoor radon measurements. The study was limited in scope and not designed to statistically represent the distribution of ambient radon concentrations for the United States. However, estimates of annual average ambient radon concentration and associated error estimates can be derived at each site since the radon samples were stratified by quarter.

ORP's Las Vegas, Nevada facility (ORP-LVF) conducted the study, but ERAMS station operators were responsible for deployment and retrieval of detectors. ORP generally relies on State personnel to operate and maintain these stations. ORP-LVF selected the short-term EIC paired with a thermoluminescent dosimeter (TLD) for this study. The short-term EIC deployed for 90 days provided adequate sensitivity to meet the data quality objectives to and from the ERAMS stations.

Since EICs respond to ionizations both from radon decay within the detector chamber and from external gamma radiation, TLDs were used to measure and correct for background gamma radiation.

At each ERAMS station, three EICs and three TLDs were placed in ventilated shelters located approximately one meter from ground level to provide uniformity. The devices were left in place for 90 days and exchanged each quarter by station operators. The quarterly results were compared and have been combined into an average annual radon concentration for each site.

The primary goal of the study, to measure outdoor ambient radon concentrations at different geographic locations in the United States, has been met with good results. For the sites examined, the mean annual outdoor concentrations ranged from a low of 0.16 at one site to a high of 0.57 pCi/L at another. The individual concentrations used to calculate mean concentrations ranged from 0.0 to 1.11 pCi/L, with a median of 0.39 pCi/L. In the study to date, field measurements using short-term EICs have been made with acceptable errors and the devices have exhibited sufficient sensitivity for measuring ambient levels of radon.

APPENDIX B

BACKGROUND INFORMATION ON RADON

Radon is a naturally-occurring, chemically inert, radioactive gas. Because radon is chemically unreactive with most materials, it is free to travel as a gas. It can move easily through very small spaces such as those between particles of soil and rock. Radon is odorless, invisible, and without taste; thus, it cannot be detected with the human senses. Radon is also moderately soluble in water and, therefore, can be absorbed by water flowing through rock or sand. Its solubility depends on the water temperature; the colder the water, the greater radon's solubility.

The two natural sources of radon, thorium and uranium, are common, naturally-occurring elements that are found in low concentrations in rock and soil. Through radioactive decay, both are constant sources of radon. Radon is produced from the radioactive decay of the element radium, which is itself a decay product of either uranium or thorium. Radioactive decay is a process in which an unstable atomic nucleus undergoes spontaneous transformation, by emission of particles or electromagnetic radiation, to form a new nucleus (decay product), which may or may not be radioactive. The level of radioactivity is measured in <u>curies</u>, where 1 curie equals 37 billion disintegrations per second. The time required for a given specific activity of an isotope to be reduced by a factor of two is called its <u>half-life</u>. A picocurie (pCi) is equal to one-trillionth of a curie. Specific activity concentrations are typically measured in picocuries per gram (in a solid) or picocuries per liter (in a gas, such as air).

Uranium-238 decays in several steps to radium-226, which decays into radon-222. Radon-222 has a half-life of 3.8 days and, therefore, has enough time to diffuse through dry, porous soils or to be transported in water for a considerable distance before it decays. Similarly, thorium-232 decays into radon-220 (a different radon isotope, also called thoron), which has a half-life of only 55 seconds. Because of its short half-life and limited ability to migrate into residences, radon-220 is usually a less important source of radon exposure to humans. The United Nations Scientific Committee on the Effects of Atomic Radiation has estimated the average exposure from indoor radon-220 decay products to be about 25 percent of that from radon-222. Only radon-222 is addressed specifically in the *Citizen's Guide* because it is the radon isotope of most concern to the public. Although radon-220, or thoron, has not been measured separately in most homes, radon mitigation will also reduce exposure to thoron.

Radon-222 is preceded in the uranium-238 decay series by radium-226, which has a half-life of 1,600 years. Radon-222 decays in several steps to form radioactive isotopes with short half-lives: polonium-218, lead-214, bismuth-214, and polonium-214. These isotope particles are commonly referred to as radon decay products, daughters, or progeny. Radon decay products are chemically reactive and can attach themselves to walls, floors, or airborne particles that are inhaled into the lungs. Unattached radon decay products also can be inhaled and, subsequently, can become deposited on lung tissue.

The four radon-222 decay products just mentioned all have half-lives of less than 30 minutes. This short half-life is significant since, once deposited on lung tissue, the radon decay products can undergo considerable decay before the action of mucus in the bronchial tubes can clear these radioactive particles. Two of the short-lived decay products, polonium-218 and polonium-214, emit alpha particles during the decay process. An alpha particle is a subatomic particle that has two protons and two neutrons and has a double positive electrical charge. It is identical to a helium nucleus.

Radon-222 is found virtually everywhere in at least small amounts because its predecessor, radium-226 (or, more distantly, uranium-238), is found in all rock and soil. In outdoor air, radon concentrations are usually less than one picocurie per liter (pCi/L) (see Appendix A for a summary of the results of EPA's Ambient Radon Field Study). Higher concentrations of radon outdoors may be observed during brief periods, such as during a temperature inversion, when a warm air mass traps a colder one beneath it. The highest individual concentration measured outdoors in EPA's Ambient Radon Field Study was 1.11 pCi/L. Indoor air concentrations, in contrast, can vary from around 0.5 pCi/L to over 2,000 pCi/L, with results from EPA's National Residential Radon Survey indicating that over 6 percent of all homes nationwide have average annual indoor radon levels above 4 pCi/L. Most indoor radon comes from the rocks and soil around a home, although other, usually less significant, sources of indoor radon are water and some construction materials.

APPENDIX C

COMMONLY USED RADON MEASUREMENT DEVICES

Activated Charcoal Adsorption Devices - Activated charcoal adsorption devices (AC) are passive devices requiring no power to function. The passive nature of the activated charcoal allows continual adsorption and desorption of radon. During the measurement period, the adsorbed radon undergoes radioactive decay. Therefore, the technique does not uniformly integrate radon concentrations during the exposure period. As with all devices that store radon, the average concentration calculated using the mid-exposure time is subject to error if the ambient radon concentration adsorbed during the first half of the sampling period is substantially higher or lower than the average over the period. For a 2- to 7-day exposure period, the lower level of detection (LLD) should be 0.5 Pci/L or less (U.S. EPA/Radon Division 1989). This level of sensitivity can normally be achieved with a counting time of up to 30 minutes. This LLD should be calculated using the results of charcoal background determinations. The coefficient of variation should not exceed 10 percent (1 sigma) at radon concentrations of 4 pCi/L or greater (U.S. EPA/Radon Division 1989). This precision should be monitored using the results of duplicate canister analyses. ACs can achieve an average coefficient of variation of less than 5 percent at concentrations of 4 pCi/L or greater.

Alpha Track Detectors - An alpha track detector (AT) consists of a small piece of plastic or film enclosed in a container with a filter-covered opening. Radon diffuses through the filter into the container and alpha particles emitted by radon and its decay products strike the detector and produce submicroscopic damage tracks. At the end of the measurement period, the detectors are returned to a laboratory. Plastic detectors are placed in a caustic solution that accentuates the damage tracks so they can be counted using a microscope or an automated counting system. The number of tracks per unit area is correlated to the radon concentration in air, using a conversion factor derived from data generated at a calibration facility. The number of tracks produced per unit time is proportional to the radon concentration, so an AT functions as a true integrating detector and measures the average concentration over the measurement period. The LLD (sensitivity) and precision of an AT system is dependent upon the tracks counted, and therefore, the area of the detector that is analyzed. With present ATs, routine counting can achieve an LLD of 1 pCi/L-month, and an LLD of 0.2 pCi/L-month may be achieved by counting additional area. The coefficient of variation (precision) should be monitored using the results of duplicate detectors. The coefficient of variation should not exceed 20 percent (1 sigma) at radon concentrations of 4 pCi/L or greater (U.S. EPA Measurement Protocols 1989).

Electret Ion Chamber Radon Detectors - Electret ion chamber radon detectors (EICs) require no power and function as true integrating detectors, measuring the average concentration during the exposure period. EICs contain a permanently charged electret (an electrostatically charged disk of Teflon[r]), which collects ions formed in the chamber by radiation emitted from radon decay products. When the device is exposed, radon diffuses into the chamber through filtered openings. Ions that are generated continuously by the decay of radon and radon decay products are drawn to the surface of the electret and reduce its surface voltage. The amount of voltage reduction is directly related to the average radon concentration present during the exposure period. There are both short-term (2- to 7-day) and long-term (1- to 12-month) EICs that are currently marketed. The thickness of the electret affects the usable measurement period. For a 7-day exposure period using a short-term EIC, as well as for a long-term EIC, the LLD (sensitivity) is about 0.3 pCi/L (U.S. EPA/Radon Division 1989). The coefficient of variation should not exceed 10 percent (1 sigma) at radon concentrations of 4 pCi/L or greater (U.S. EPA/Radon Division 1989). This precision should be monitored by using results of duplicate detector analyses.

Continuous Radon Monitors - There are three types of continuous radon monitors (CRs). In the first type, ambient air is sampled for radon in a scintillation cell after passing through a filter that removes radon decay products and dust. Alpha particles (produced by radon decays) strike the zinc sulphide coating of the cell, yielding scintillations that are detected by a photo-multiplier tube in the detector. A second type of CR operates as an ionization chamber. Radon in the ambient air diffuses into the chamber through a filtered area so that the radon concentration in the chamber follows the radon concentration in the ambient air with some small time lag. The third type of CR functions by allowing ambient air to diffuse through a filter into a detection chamber. As radon decays, the alpha particles are counted using a solid state silicon detector. Most CRs are capable of an LLD (sensitivity) of 0.5 pCi/L or less (U.S. EPA/Radon Division 1989). Special cells are available for some CRs that have LLDs of 0.1 pCi/L. The precision of most CRs can achieve a coefficient of variation of less than 10 percent (1 sigma) at 4 pCi/L or greater (U.S. EPA/Radon Division 1989).

APPENDIX D

COVERAGE OF RADON TESTING POLICY

EPA's estimation of the housing stock and residents covered by its radon testing policy required the determination of how many units were covered in each category of structures (e.g. single-family homes) and the occupancy rates of these units. It was assumed that the units covered would be homes that were intended for regular (year-round) use. Units that were seasonally or occasionally used (by Census definitions), or could not be linked to regular usage (the Census classifies them as "other" vacancies) were assumed to not be covered by EPA's testing policy.

The coverage estimates are based on 1990 Census data. Many of these data were available through 1991 Census press releases. The remainder was made available to EPA from internal Census documents that are normally used to answer specific inquiries.

Adjustments of the Census data were necessary because the specific statistics needed were not available (e.g., number of group quarters units), or because EPA's testing policy only covered a segment of a housing group (e.g., units below the third floor in multi-unit structures.) Some statistics were available nationally; others had to be constructed from State-level data. The results of the analysis appear in Exhibit D-1. A summary of how the estimates of housing units and residents covered for each major housing group were developed follows.

<u>Housing Units</u>

- **Single-Family Homes**: The 1990 Census collected data on the number of single-family homes. This figure was then adjusted to reflect only homes intended for year-round use.[1]

- **Mobile Homes**: The number of mobile homes placed on permanent foundations was assumed to be five percent of all mobile homes. This assumption was based on a review of new construction reports published by the Census and information collected from trade associations.

- **Multi-Units**: The 1990 Census collected data on the number of multi-units. This figure was then adjusted to reflect only multi-units intended for year-round use, using the same approach used for single-family homes. The number of apartments in multi-unit structures that are below the third floor was assumed to be two-thirds of the total number of apartments in multi-unit structures. This was derived from an assessment of the relationship between multi-unit structures and numbers of stories in these buildings.

- **Group Quarters**: The number of group quarters was calculated by dividing the number of people living in group quarters (also taken from the 1990 Census data) by the national

[1]A vacancy adjustment factor was calculated by dividing the number of housing units used occasionally by the total number of housing units and adding that figure to the adjustment factor for homes occupied by residents that "usually reside elsewhere" (URE) (e.g., travellers). The Census Bureau provided a national URE factor of approximately 2 percent. The vacancy adjustment factor was then used to determine the number of units that were <u>not</u> intended for regular use.

occupancy rate for multi-units.[2] An assumption was made that these structures have comparable occupancy rates given their similarity. The number of group quarters below the third floor was assumed to be two-thirds the total number of group quarters.

Residents in Housing Units

The population affected by the testing and mitigation program includes all residents of single-family homes, multi-units, mobile homes, and group quarters that should test.

- **Single-Family Homes**: Based on information from the Census Bureau, the average occupancy rate of single-family homes that are intended for year-round use was determined to be about 2.8 persons per home. This occupancy rate was then multiplied by the number of single-family homes intended for year-round use.

- **Mobile Homes and Multi-Units**: The occupancy rate for mobile homes and multi-units was calculated by taking the total national population, subtracting out the number of people living in single-family homes and group quarters, and dividing the remainder by the number of mobile homes and multi-units. This occupancy rate was then multiplied by the number of mobile homes and multi-units that should test.

- **Group Quarters**: The number of people residing in group quarters was taken from State-by-State results from the 1990 Census. The number was multiplied by two-thirds to calculate the number of residents in group quarters that should test. This was done based on the assumption that group quarters have the same distribution of stories and building geometry as multi-unit structures.

[2] The national occupancy rate of multi-units was assumed to be comparable to the occupancy rate of group quarters.

EXHIBIT D-1 ESTIMATION OF 1990 U.S. POPULATION IN DIFFERENT TYPES OF HOUSING
REQUIRING RADON TESTING AND POTENTIAL MITIGATION BY STATE
(IN THOUSANDS)

STATE	POPULATION	SINGLE-FAMILY HOMES	RESIDENTS OF SINGLE-FAMILY HOMES	MOBILE HOMES	RESIDENTS OF MOBILE HOMES	MOBILE HOMES WITH PERMANENT FOUNDATIONS	RESIDENTS OF MOBILE HOMES WITH PERMANENT FOUNDATIONS	MULTI-UNITS
Alabama	4,041	1,117	3,128	230	389	11	19	254
Alaska	550	127	356	22	46	1	2	61
Arizona	3,665	901	2,524	253	428	13	21	375
Arkansas	2,351	699	1,957	136	173	7	9	127
California	29,760	6,669	18,672	654	1,653	33	83	3,437
Colorado	3,294	910	2,548	96	135	5	7	378
Connecticut	3,287	786	2,201	30	60	1	3	458
Delaware	666	179	501	34	57	2	3	52
District of Columbia	607	103	289	3	5	0	0	165
Florida	12,938	3,069	8,594	748	1,213	37	61	1,741
Georgia	6,478	1,656	4,635	317	591	16	30	578
Hawaii	1,108	224	628	6	13	0	1	139
Idaho	1,007	272	761	55	114	3	6	54
Illinois	11,431	2,644	7,405	184	395	9	20	1,561
Indiana	5,544	1,572	4,400	170	281	8	14	422
Iowa	2,777	842	2,357	66	80	3	4	198
Kansas	2,478	761	2,130	76	79	4	4	178
Kentucky	3,685	1,001	2,802	193	332	10	17	262
Lousiana	4,220	1,119	3,132	209	383	10	19	323
Maine	1,228	324	906	56	93	3	5	107
Maryland	4,781	1,275	3,569	54	110	3	5	483
Massachusetts	6,016	1,251	3,503	48	103	2	5	1,033
Michigan	9,295	2,584	7,234	266	512	13	26	696
Minnesota	4,375	1,199	3,358	103	188	5	9	403
Mississippi	2,573	701	1,963	144	285	7	14	129
Missouri	5,117	1,476	4,134	174	234	9	12	449
Montana	799	227	636	54	71	3	4	52
Nebraska	1,578	477	1,334	41	50	2	2	119
Nevada	1,202	252	705	72	140	4	7	173
New Hampshire	1,109	278	778	36	68	2	3	122
New Jersey	7,730	1,773	4,963	73	165	4	8	1,067
New Mexico	1,515	393	1,101	106	200	5	10	98
New York	17,990	3,070	8,597	288	670	14	34	3,509
North Carolina	6,629	1,799	5,037	429	679	21	34	434
North Dakota	639	175	489	28	40	1	2	61
Ohio	10,847	2,956	8,278	239	429	12	21	1,050
Oklahoma	3,146	998	2,795	138	100	7	5	216
Oregon	2,842	760	2,129	139	237	7	12	240
Pennsylvania	11,882	3,370	9,437	304	482	15	24	1,018
Rhode Island	1,003	219	612	8	17	0	1	167
South Carolina	3,487	880	2,465	239	466	12	23	226
South Dakota	696	197	552	33	47	2	2	48
Tennessee	4,877	1,368	3,831	201	310	10	16	392
Texas	16,987	4,410	12,349	604	1,113	30	56	1,700
Utah	1,723	394	1,103	39	134	2	7	132
Vermont	563	144	404	23	42	1	2	53
Virginia	6,187	1,683	4,713	175	308	9	15	545
Washington	4,867	1,258	3,521	197	357	10	18	480
West Virginia	1,793	530	1,485	122	156	6	8	91
Wisconsin	4,892	1,262	3,534	117	239	6	12	484
Wyoming	454	126	354	33	47	2	2	30
TOTALS	248,710	62,461	174,891	8,067	14,526	403	726	26,569

EXHIBIT D-1 ESTIMATION OF 1990 U.S. POPULATION IN DIFFERENT TYPES OF HOUSING REQUIRING RADON TESTING AND POTENTIAL MITIGATION BY STATE (IN THOUSANDS)

STATE	RESIDENTS OF MULTI-UNITS	MULTI-UNITS THAT SHOULD TEST	RESIDENTS OF MULTI-UNITS THAT SHOULD TEST	GROUP QUARTERS	RESIDENTS OF GROUP QUARTERS	GROUP QUARTERS THAT SHOULD TEST	RESIDENTS OF GROUP QUARTERS THAT SHOULD TEST	ALL UNITS THAT SHOULD TEST	RESIDENTS OF ALL UNITS THAT SHOULD TEST
Alabama	431	170	289	47	92	31	62	1,330	3,498
Alaska	127	41	85	10	21	7	14	176	457
Arizona	633	251	424	41	81	27	54	1,192	3,023
Arkansas	162	85	109	29	58	20	39	811	2,113
California	8,683	2,303	5,818	380	752	254	504	9,258	25,076
Colorado	532	253	356	40	79	27	53	1,195	2,965
Connecticut	925	307	620	51	101	34	68	1,128	2,891
Delaware	87	35	59	10	20	7	13	222	576
District of Columbia	272	111	182	21	42	14	28	228	499
Florida	2,823	1,166	1,892	155	307	104	206	4,377	10,752
Georgia	1,078	388	722	88	174	59	116	2,118	5,504
Hawaii	425	93	284	19	38	13	25	330	939
Idaho	110	36	74	11	21	7	14	318	855
Illinois	3,344	1,046	2,241	145	287	97	192	3,796	9,857
Indiana	701	283	469	82	162	55	109	1,918	4,992
Iowa	240	132	161	50	100	34	67	1,011	2,588
Kansas	185	120	124	42	83	28	55	912	2,314
Kentucky	451	175	302	51	101	34	68	1,220	3,188
Lousiana	592	216	397	57	113	38	75	1,384	3,623
Maine	186	72	125	19	37	13	25	411	1,061
Maryland	988	323	662	58	114	39	76	1,639	4,313
Massachusetts	2,197	692	1,472	108	214	73	144	2,018	5,123
Michigan	1,338	466	896	107	212	72	142	3,135	8,298
Minnesota	716	270	480	59	118	40	79	1,515	3,926
Mississippi	256	86	171	35	70	24	47	818	2,195
Missouri	604	301	405	73	145	49	97	1,835	4,648
Montana	69	35	46	12	24	8	16	273	701
Nebraska	147	80	98	24	48	16	32	574	1,467
Nevada	333	116	223	12	24	8	16	379	951
New Hampshire	230	82	154	16	32	11	22	373	958
New Jersey	2,430	715	1,628	87	171	58	115	2,549	6,714
New Mexico	185	66	124	15	29	10	19	474	1,254
New York	8,178	2,351	5,479	275	545	185	365	5,620	14,475
North Carolina	687	291	461	113	224	76	150	2,187	5,682
North Dakota	86	41	57	12	24	8	16	225	585
Ohio	1,879	703	1,259	132	261	88	175	3,760	9,734
Oklahoma	157	145	105	47	94	32	63	1,182	2,968
Oregon	411	161	275	33	66	22	44	951	2,460
Pennsylvania	1,614	682	1,081	176	348	118	233	4,186	10,776
Rhode Island	336	112	225	19	39	13	26	344	864
South Carolina	439	151	294	59	117	39	78	1,083	2,861
South Dakota	70	32	47	13	26	9	17	240	619
Tennessee	606	263	406	65	129	44	87	1,685	4,340
Texas	3,131	1,139	2,098	199	393	133	264	5,713	14,766
Utah	457	89	306	15	29	10	19	494	1,435
Vermont	95	35	64	11	22	7	15	188	484
Virginia	957	365	641	106	209	71	140	2,128	5,510
Washington	868	322	582	61	121	41	81	1,630	4,202
West Virginia	116	61	78	19	37	12	25	610	1,595
Wisconsin	985	324	660	67	134	45	90	1,637	4,296
Wyoming	42	20	28	5	10	3	7	152	391
TOTALS	52,596	17,801	35,239	3,383	6,698	2,267	4,487	82,932	215,344

APPENDIX E

RISK ANALYSIS METHODOLOGY

The assessment of reductions in risks that result from the public following the testing procedure advice in the final revised *Citizen's Guide* ("Testing Option C" in Chapter 3) for alternative action levels of 2 pCi/L, 3 pCi/L, and 4 pCi/L had two major components:

- Estimation of the number of residents in homes that have testing results above the action level and should mitigate their radon levels.

- Calculation of the risk reductions that should result when these mitigations occur.

Each of these components of the analysis is discussed in the sections that follow.

Residential Population that Should Mitigate

To determine the size of the residential population that would be mitigating their homes, it was necessary to (1) estimate the residential coverage of EPA's radon testing policy and (2) predict the testing results that this group would have if it followed EPA's advice in the revised *Citizen's Guide*. Appendix D explains how the Agency estimated the population covered by EPA's testing policy. Based on 1990 Census data, EPA estimates that about 215 million people are covered by the testing policy. The remainder of this discussion explains how EPA predicted the results of this group's radon testing (using Option C).

The public is advised to conduct an initial short-term test. If the results are above the action level, Option C gives the public the choice of (1) conducting a follow-up long-term test and fixing the house if the results are above the action level (like Option A); or 2) conducting a second short-term test and fixing the house if the average of the two results is above the action level (like Option B). EPA believes that under Option C, the public will actually pursue Option A testing about 9 percent of the time and Option B testing about 91 percent of the time. EPA used a statistical model to predict the radon test results from using Testing Option C for alternative action levels by independently looking at the predicted outcomes for Option A and Option B for each action level (Chmelynski 1990). EPA weighted the results by the expected usage of each testing approach by the public and then combined them to provide Option C results.

The statistical model used to predict testing results provided the joint probability distribution for the results of short-term testing given the actual long-term radon levels houses were likely to have. Exhibits E-1 through E-3 show these distributions for each action level examined. Each exhibit provides results of Option A, Option B, and Option C, which is the combination of the preceding two options. Summations of the appropriate portions of each of these distributions leads to estimates of the fraction of homes that will need to be fixed. Multiplying these fractions by the residential population covered in EPA's testing policy yields the number of persons affected. For instance, the fraction of homes that have less than 1 pCi/L of radon and that will have false positive radon testing results under Option B is the summation of the exhibit entries in the row "0-1" that begins from "4-6" to ">20." This fraction multiplied by 215 million people provides the population that lives in homes that are below 1 pCi/L and that would get false positive results, if all homeowners used Option B to test their homes for radon.

EPA generated the distribution tables for each action level based on testing devices used, number of home floors, and testing conditions.[1] EPA assumed there was an equal chance of homeowners using short-term devices like charcoal canisters and alpha track detectors. For charcoal canisters, EPA assumed that all tests were done under closed-house conditions.[2] For alpha track detectors (ATDs), EPA assumed that 50 percent of tests would occur under open-house conditions and that 50 percent would occur under closed-house conditions. Based on data from the *1990 Census Data* and the *Characteristics of New Housing: 1990 Current Construction Reports*, the weights for single floor vs. multi-floor homes were determined to be 46 and 54 percent, respectively. (The same approach that EPA used to derive the mitigation classification rates in Chapter 3 was also used here, except the focus was on options that had different action levels using the same Option C testing approach.)

Risk Reductions Due to Mitigation

EPA assumed for the Chapter 5 analysis that all homeowners of houses testing above the action levels in long- or short-term testing installed and operated mitigation systems. The Agency used the radon testing results shown in the first three exhibits to develop estimates of homes at various radon levels that mitigated, as well as the exposure reductions they would receive, for each of the selected pCi/L intervals in these exhibits. To estimate the exposure reductions received in each interval, EPA subtracted the level of radon existing after mitigation from the radon levels estimated to exist within each interval before mitigation. The average pre-mitigation radon levels in the intervals were taken from results of the EPA National Residential Radon Survey. Exhibit E-4 provides these average levels. Based on the mitigation research and experience that is summarized in Chapter 4, EPA assumed that homes that were above 2 pCi/L would have their levels reduced to 2 pCi/L (on average) and homes below that level would not have any radon reductions.[3] The same type of approach was used to estimate the risk reductions that were not gained ("lives lost") when mitigation of homes above the action levels did not occur due to false negative testing results.

To facilitate the assessment of risk reductions that result from exposure reductions, EPA developed multipliers that provided the lung cancer deaths averted per million persons receiving the exposure reduction in each interval. The multipliers were based on EPA's central estimate of the risk factor (i.e., that there will be 43.2 lung cancer deaths annually per million persons annually exposed to one pCi/L). Exposure reductions (in pCi/L) for each interval were multiplied by the risk factor to arrive at each interval's own multiplier. Exhibit E-4 shows the multipliers that were used in all the analyses of the action levels.

EPA independently calculated risk reductions gained (or lost) for homes that had false positive test results, true positive test results, and false negative test results. Exhibits E-5 through E-7 show these results for each action level. The final results that appear in Chapter 5 are the Option C results in each exhibit. The intermediate results for Options A and B are also shown in the exhibits. These results assume that the entire testing population (215 million people) follows that

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[1]Distribution tables were generated using An evaluation of the performance of alternative short-term radon testing procedures in homes with pending real estate transactions. Chmelynski, H. 1990. Submitted to EPA.

[2]For a more detailed explanation of radon tests, please refer to Chapter 3.

[3]There will often be some risk reduction that will occur for homes below 2 pCi/L. Due to the lack of data on exactly how much reduction, however, EPA made a conservative assumption that there would be no reduction.

testing option. The results from Options A and B are then weighted by the portion of the population that follows each option to derive the results for Option C. Exhibit E-8 helps explain the reasons for the differences in the levels of risk reduction that can occur given the exposure distribution that exists in the U.S. housing stock.

Exhibit E-1
Weighted Testing Distributions
at Action Level 4pCi/L

OPTION A

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.09305886	0.03151984	0.01374314	0.01085996	0.00384689	0.00168459	0.00187647	0.00031058	0.61979767
1-2	0.07260808	0.06399632	0.03126357	0.01589892	0.01373066	0.00507591	0.00221464	0.00232588	0.00028592	0.20739993
2-3	0.00935460	0.01951012	0.01472083	0.00945063	0.00979078	0.00421989	0.00201153	0.00231958	0.00032199	0.07169998
3-4	0.00196702	0.00694166	0.00740183	0.00588678	0.00734195	0.00372795	0.00196585	0.00254703	0.00041995	0.03820005
4-6	0.00068686	0.00364974	0.00532080	0.00529788	0.00824628	0.00515586	0.00312115	0.00480042	0.00102097	0.03730000
6-8	0.00005831	0.00047387	0.00095279	0.00121286	0.00245525	0.00197302	0.00142079	0.00274733	0.00080572	0.01209997
8-10	0.00001098	0.00011760	0.00028938	0.00043318	0.00106341	0.00103796	0.00086413	0.00206755	0.00081574	0.00669997
10-20	0.00000251	0.00003546	0.00010599	0.00018565	0.00055858	0.00068153	0.00068512	0.00231919	0.00162592	0.00619998
>20	0	0.00000019	0.00000091	0.00000217	0.00000943	0.00001679	0.00002330	0.00015659	0.00039030	0.00059971
TOTAL	0.54758572	0.18778386	0.09157597	0.05211125	0.05405634	0.02573583	0.01399113	0.02116006	0.00599713	0.99999731

OPTION B

Two rounds of testing for positives (one round of testing for negatives)

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.09305886	0.03930476	0.01827882	0.00393081	0.00132012	0.00051290	0.00045174	0.00004232	0.61979768
1-2	0.07260808	0.06399632	0.03809300	0.02211290	0.00664640	0.00236203	0.00087742	0.00066808	0.00003566	0.20739994
2-3	0.00935460	0.01951012	0.01772150	0.01379046	0.00636274	0.00278552	0.00114636	0.00096730	0.00006135	0.07169998
3-4	0.00196702	0.00694166	0.00874512	0.00871796	0.00570512	0.00314840	0.00146495	0.00139973	0.00011005	0.03820005
4-6	0.00068686	0.00364974	0.00613777	0.00771167	0.00703149	0.00529483	0.00297480	0.00344802	0.00036477	0.03729999
6-8	0.00005831	0.00047387	0.00106081	0.00165693	0.00199744	0.00224083	0.00164993	0.00256784	0.00039397	0.01209996
8-10	0.00001098	0.00011760	0.00031517	0.00055988	0.00075293	0.00113674	0.00106112	0.00225052	0.00049497	0.00669995
10-20	0.00000251	0.00003546	0.00011330	0.00022744	0.00031344	0.00062719	0.00077200	0.00278277	0.00132583	0.00619996
>20	0	0.00000019	0.00000094	0.00000241	0.00000277	0.00000856	0.00001597	0.00016090	0.00040793	0.00059972
TOTAL	0.54758572	0.18778386	0.11149240	0.07305851	0.03274317	0.01892425	0.01047548	0.01469695	0.00323690	0.99999728

OPTION C

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46456937	0.09305886	0.03860412	0.01787061	0.00357704	0.00120131	0.00046674	0.00041108	0.00003851	0.61979768
1-2	0.07260808	0.06612329	0.03747835	0.02155364	0.00604823	0.00214945	0.00079846	0.00060795	0.00003245	0.20739994
2-3	0.00935460	0.01951012	0.01913118	0.01339987	0.00579009	0.00253482	0.00104318	0.00088024	0.00005582	0.07169998
3-4	0.00196702	0.00694166	0.00862422	0.00990340	0.00519166	0.00286504	0.00133310	0.00127375	0.00010015	0.03820005
4-6	0.00068686	0.00364974	0.00606425	0.00749443	0.00840968	0.00481829	0.00270707	0.00313769	0.00033194	0.03729999
6-8	0.00005831	0.00047387	0.00105109	0.00161697	0.00181767	0.00288534	0.00150144	0.00233673	0.00035851	0.01209996
8-10	0.00001098	0.00011760	0.00031285	0.00054848	0.00068517	0.00103444	0.00149201	0.00204798	0.00045042	0.00669996
10-20	0.00000251	0.00003546	0.00011264	0.00022367	0.00028523	0.00057074	0.00070252	0.00306066	0.00120650	0.00619997
>20	0	0.00000019	0.00000093	0.00000239	0.00000252	0.00000779	0.00001453	0.00014642	0.00042490	0.00059972
TOTAL	0.54925778	0.18991083	0.11137967	0.07261350	0.03180731	0.01806726	0.01005908	0.01390256	0.00299925	0.99999728

Exhibit E-2
Weighted Testing Distributions
at Action Level 3pCi/L

OPTION A

Average	Short-term Measurement (pCi/L)									
(pCi/L)	0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.09305886	0.03151984	0.01374314	0.01085996	0.00384689	0.00168459	0.00187647	0.00031058	0.61979767
1-2	0.07260808	0.06399632	0.03126357	0.01589892	0.01373066	0.00507591	0.00221464	0.00232588	0.00028592	0.20739993
2-3	0.00935460	0.01951012	0.01472083	0.00945063	0.00979078	0.00421989	0.00201153	0.00231958	0.00032199	0.07169998
3-4	0.00196702	0.00694166	0.00740183	0.00588678	0.00734195	0.00372795	0.00196585	0.00254703	0.00041995	0.03820005
4-6	0.00068686	0.00364974	0.00532080	0.00529788	0.00824628	0.00515586	0.00312115	0.00480042	0.00102097	0.03730000
6-8	0.00005831	0.00047387	0.00095279	0.00121286	0.00245525	0.00197302	0.00142079	0.00274733	0.00080572	0.01209997
8-10	0.00001098	0.00011760	0.00028938	0.00043318	0.00106341	0.00103796	0.00086413	0.00206755	0.00081574	0.00669997
10-20	0.00000251	0.00003546	0.00010599	0.00018565	0.00055858	0.00068153	0.00068512	0.00231919	0.00162592	0.00619998
>20	0	0.00000019	0.00000091	0.00000217	0.00000943	0.00001679	0.00002330	0.00015659	0.00039030	0.00059971
TOTAL	0.54758572	0.18778386	0.09157597	0.05211125	0.05405634	0.02573583	0.01399113	0.02116006	0.00599713	0.99999731

OPTION B

Two rounds of testing for positives (one round of testing for negatives)

Average	Short-term Measurement (pCi/L)									
(pCi/L)	0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.10153744	0.04280681	0.00575363	0.00445658	0.00136478	0.00050742	0.00043364	0.00003993	0.61979758
1-2	0.07260808	0.06937046	0.04403728	0.00917330	0.00812751	0.00251395	0.00088221	0.00065262	0.00003448	0.20739994
2-3	0.00935460	0.02100491	0.02111349	0.00693803	0.00806765	0.00302242	0.00117354	0.00096456	0.00006072	0.07169996
3-4	0.00196702	0.00739270	0.01041371	0.00467816	0.00729849	0.00342878	0.00150674	0.00140473	0.00010969	0.03820005
4-6	0.00068686	0.00384442	0.00716892	0.00401302	0.00893349	0.00574629	0.00306243	0.00347800	0.00036654	0.03730000
6-8	0.00005831	0.00049249	0.00119839	0.00074225	0.00251960	0.00241887	0.00169167	0.00258330	0.00039505	0.01209997
8-10	0.00001098	0.00012124	0.00034771	0.00021113	0.00094417	0.00122360	0.00108561	0.00225984	0.00049565	0.00669997
10-20	0.00000251	0.00003631	0.00012226	0.00006876	0.00038884	0.00067140	0.00078679	0.00279025	0.00133281	0.00619996
>20	0	0.00000020	0.00000097	0.00000038	0.00000343	0.00000920	0.00001633	0.00016108	0.00040808	0.00059970
TOTAL	0.54758572	0.20380020	0.12720958	0.03157871	0.04073978	0.02039933	0.01071278	0.01472806	0.00324298	0.99999718

OPTION C

Average	Short-term Measurement (pCi/L)									
(pCi/L)	0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46580626	0.10077437	0.04179098	0.00523581	0.00405548	0.00124195	0.00046175	0.00039461	0.00003633	0.61979759
1-2	0.07260808	0.07244466	0.04288765	0.00834770	0.00739603	0.00228769	0.00080281	0.00059389	0.00003138	0.20739994
2-3	0.00935460	0.02087038	0.02306845	0.00631361	0.00734156	0.00275040	0.00106792	0.00087775	0.00005525	0.07169996
3-4	0.00196702	0.00735210	0.01014264	0.00622718	0.00664162	0.00312018	0.00137113	0.00127831	0.00009982	0.03820005
4-6	0.00068686	0.00382690	0.00700259	0.00365185	0.01061731	0.00522912	0.00278681	0.00316498	0.00033355	0.03730000
6-8	0.00005831	0.00049081	0.00117629	0.00067544	0.00229283	0.00315652	0.00153942	0.00235081	0.00035950	0.01209997
8-10	0.00001098	0.00012092	0.00034246	0.00019213	0.00085920	0.00111348	0.00155328	0.00205645	0.00045104	0.00669997
10-20	0.00000251	0.00003623	0.00012080	0.00006257	0.00035384	0.00061098	0.00071597	0.00308416	0.00121285	0.00619996
>20	0	0.00000020	0.00000096	0.00000035	0.00000312	0.00000837	0.00001486	0.00014658	0.00042523	0.00059970
TOTAL	0.55049467	0.20591660	0.12653285	0.03070668	0.03956104	0.01951874	0.01031401	0.01394758	0.00300498	0.99999719

Exhibit E-3
Weighted Testing Distributions
at Action Level 2pCi/L

OPTION A

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.09305886	0.03151984	0.01374314	0.01085996	0.00384689	0.00168459	0.00187647	0.00031058	0.61979767
1-2	0.07260808	0.06399632	0.03126357	0.01589892	0.01373066	0.00507591	0.00221464	0.00232588	0.00028592	0.20739993
2-3	0.00935460	0.01951012	0.01472083	0.00945063	0.00979078	0.00421989	0.00201153	0.00231958	0.00032199	0.07169998
3-4	0.00196702	0.00694166	0.00740183	0.00588678	0.00734195	0.00372795	0.00196585	0.00254703	0.00041995	0.03820005
4-6	0.00068686	0.00364974	0.00532080	0.00529788	0.00824628	0.00515586	0.00312115	0.00480042	0.00102097	0.03730000
6-8	0.00005831	0.00047387	0.00095279	0.00121286	0.00245525	0.00197302	0.00142079	0.00274733	0.00080572	0.01209997
8-10	0.00001098	0.00011760	0.00028938	0.00043318	0.00106341	0.00103796	0.00086413	0.00206755	0.00081574	0.00669997
10-20	0.00000251	0.00003546	0.00010599	0.00018565	0.00055858	0.00068153	0.00068512	0.00231919	0.00162592	0.00619998
>20	0	0.00000019	0.00000091	0.00000217	0.00000943	0.00001679	0.00002330	0.00015659	0.00039030	0.00059971
TOTAL	0.54758572	0.18778386	0.09157597	0.05211125	0.05405634	0.02573583	0.01399113	0.02116006	0.00599713	0.99999731

OPTION B

Two rounds of testing for positives (one round of testing for negatives)

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46289731	0.12680067	0.01613261	0.00694534	0.00476502	0.00133799	0.00048029	0.00040174	0.00003646	0.61979747
1-2	0.07260808	0.08653552	0.02265543	0.01235090	0.00916006	0.00255170	0.00086859	0.00063603	0.00003357	0.20739993
2-3	0.00935460	0.02569338	0.01237851	0.00971940	0.00923427	0.00311838	0.00117925	0.00096163	0.00006051	0.07169997
3-4	0.00196702	0.00877038	0.00597427	0.00659872	0.00831560	0.00353820	0.00151918	0.00140689	0.00010977	0.03820007
4-6	0.00068686	0.00441416	0.00364563	0.00561705	0.01005635	0.00591306	0.00309478	0.00350249	0.00036960	0.03730000
6-8	0.00005831	0.00054525	0.00048325	0.00103758	0.00281113	0.00247601	0.00170199	0.00259007	0.00039636	0.01209999
8-10	0.00001098	0.00013130	0.00011404	0.00029464	0.00104921	0.00125052	0.00109080	0.00226217	0.00049627	0.00669998
10-20	0.00000251	0.00003855	0.00003140	0.00009533	0.00042971	0.00068433	0.00078908	0.00279173	0.00133729	0.00619997
>20	0	0.00000020	0.00000012	0.00000053	0.00000379	0.00000941	0.00001640	0.00016107	0.00040815	0.00059970
TOTAL	0.54758572	0.25292944	0.06141529	0.04265953	0.04582517	0.02087963	0.01074041	0.01471386	0.00324803	0.99999711

OPTION C

Average (pCi/L)	Short-term Measurement (pCi/L) 0-1	1-2	2-3	3-4	4-6	6-8	8-10	10-20	>20	TOTAL
0-1	0.46864304	0.12376390	0.01468067	0.00632026	0.00433617	0.00121757	0.00043707	0.00036559	0.00003318	0.61979749
1-2	0.07260808	0.09087859	0.02061644	0.01123932	0.00833566	0.00232205	0.00079042	0.00057879	0.00003055	0.20739993
2-3	0.00935460	0.02513689	0.01511961	0.00884466	0.00840318	0.00283772	0.00107312	0.00087508	0.00005507	0.07169997
3-4	0.00196702	0.00860579	0.00543659	0.00864105	0.00756720	0.00321976	0.00138245	0.00128027	0.00009989	0.03820007
4-6	0.00068686	0.00434537	0.00331752	0.00511152	0.01211798	0.00538088	0.00281625	0.00318726	0.00033633	0.03730000
6-8	0.00005831	0.00053883	0.00043975	0.00094420	0.00255813	0.00329427	0.00154881	0.00235697	0.00036069	0.01209998
8-10	0.00001098	0.00013007	0.00010378	0.00026812	0.00095478	0.00113797	0.00158405	0.00205857	0.00045161	0.00669998
10-20	0.00000251	0.00003827	0.00002858	0.00008675	0.00039103	0.00062274	0.00071806	0.00309506	0.00121693	0.00619997
>20	0	0.00000020	0.00000011	0.00000048	0.00000345	0.00000856	0.00001492	0.00014658	0.00042537	0.00059970
TOTAL	0.55333145	0.25343794	0.05974308	0.04145639	0.04466761	0.02004156	0.01036520	0.01394420	0.00300966	0.99999713

Exhibit E-4
Radon Exposure and Deaths Per Million
In Each Interval

pCi/l Interval	Arithmetic Mean Exposure In Interval	Deaths per Million in Interval
0-1	0.440	NONE
1-2	1.400	NONE
2-3	2.440	19
3-4	3.430	62
4-6	4.942	127
6-8	6.900	212
8-10	8.950	300
10-20	12.890	470
> 20	29.250	1,177

Exhibit E-5
Risk Calculations for Existing Homes Using
Distribution for All Homes that Should Test
at Action Level 4pCi/L

Option A

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False
1-2	FP/TN	0.2074	NONE	NONE	0	Positive
2-3	FP/TN	0.0717	NONE	NONE	0	0
3-4	FP/TN	0.0382	NONE	NONE	0	
4-6	TP	0.0223	2	127	612	
6-8	TP	0.0094	2	212	429	True
8-10	TP	0.0058	2	300	378	Positive
10-20	TP	0.0059	2	470	595	2,164
>20	TP	0.0006	2	1,177	151	
4-6	FN	0.0150	NONE	127	(409)	
6-8	FN	0.0027	NONE	212	(123)	False
8-10	FN	0.0009	NONE	300	(55)	Negative
10-20	FN	0.0003	NONE	470	(33)	(622)
>20	FN	0.0000	NONE	1,177	(1)	

Option B

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False
1-2	FP/TN	0.2074	NONE	NONE	0	Positive
2-3	FP/TN	0.0717	2	19	46	204
3-4	FP/TN	0.0382	2	62	157	
4-6	TP	0.0191	2	127	523	
6-8	TP	0.0089	2	212	403	True
8-10	TP	0.0057	2	300	368	Positive
10-20	TP	0.0058	2	470	590	2,036
>20	TP	0.0006	2	1,177	151	
4-6	FN	0.0182	NONE	127	(498)	
6-8	FN	0.0032	NONE	212	(148)	False
8-10	FN	0.0010	NONE	300	(65)	Negative
10-20	FN	0.0004	NONE	470	(38)	(750)
>20	FN	0.0000	NONE	1,177	(1)	

Exhibit E-5 (continued)

Option C

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False
1-2	FP/TN	0.2074	NONE	NONE	0	Positive
2-3	FP/TN	0.0717	NONE/2**	17	42	185
3-4	FP/TN	0.0382	NONE/2**	56	143	
4-6	TP	0.0194	2	127	531	
6-8	TP	0.0089	2	212	406	True
8-10	TP	0.0057	2	300	369	Positive
10-20	TP	0.0058	2	470	590	2,047
>20	TP	0.0006	2	1,177	151	
4-6	FN	0.0179	NONE	127	(490)	
6-8	FN	0.0032	NONE	212	(146)	False
8-10	FN	0.0010	NONE	300	(64)	Negative
10-20	FN	0.0004	NONE	470	(38)	(738)
>20	FN	0.0000	NONE	1,177	(1)	

*True Positive (TP), False Positive (FP), True Negative (TN), and False Negative (FN). Assumes 100% testing and mitigation. Column entries may not sum due to rounding of estimates.

**There is no radon reduction level for TN homes, there is a radon reduction level of 2 for homes that test FP.

Exhibit E-6
Risk Calculations for Existing Homes Using
Distribution for All Homes that Should Test
at Action Level 3pCi/L

Option A

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	FP/TN	0.0717	NONE	NONE	0	0
3-4	TP	0.0219	2	62	291	True Positive
4-6	TP	0.0276	2	127	757	
6-8	TP	0.0106	2	212	484	
8-10	TP	0.0063	2	300	406	
10-20	TP	0.0061	2	470	614	2,703
>20	TP	0.0006	2	1,177	152	False Negative
3-4	FN	0.0163	NONE	62	(217)	
4-6	FN	0.0097	NONE	127	(264)	
6-8	FN	0.0015	NONE	212	(68)	
8-10	FN	0.0004	NONE	300	(27)	
10-20	FN	0.0001	NONE	470	(15)	(591)
>20	FN	0.0000	NONE	1,177	(0)	

Option B

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	FP/TN	0.0717	2	19	83	83
3-4	TP	0.0184	2	62	245	True Positive
4-6	TP	0.0256	2	127	701	
6-8	TP	0.0096	2	212	438	
8-10	TP	0.0060	2	300	389	
10-20	TP	0.0060	2	470	605	2,529
>20	TP	0.0006	2	1,177	152	False Negative
3-4	FN	0.0198	NONE	62	(263)	
4-6	FN	0.0117	NONE	127	(320)	
6-8	FN	0.0025	NONE	212	(114)	
8-10	FN	0.0007	NONE	300	(45)	
10-20	FN	0.0002	NONE	470	(23)	(765)
>20	FN	0.0000	NONE	1,177	(0)	

Exhibit E-6 (continued)

Option C

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive 75
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	FP/TN	0.0717	NONE/2**	17	75	
3-4	TP	0.0187	2	62	249	True Positive 2,544
4-6	TP	0.0258	2	127	706	
6-8	TP	0.0097	2	212	442	
8-10	TP	0.0060	2	300	390	
10-20	TP	0.0060	2	470	606	
>20	TP	0.0006	2	1,177	152	False Negative (750)
3-4	FN	0.0195	NONE	62	(259)	
4-6	FN	0.0115	NONE	127	(315)	
6-8	FN	0.0024	NONE	212	(109)	
8-10	FN	0.0007	NONE	300	(43)	
10-20	FN	0.0002	NONE	470	(23)	
>20	FN	0.0000	NONE	1,177	(0)	

*True Positive (TP), False Positive (FP), True Negative (TN), and False Negative (FN). Assumes 100% testing and mitigation. Column entries may not sum due to rounding of estimates.

**There is no radon reduction level for TN homes, there is a radon reduction level of 2 for homes that test FP.

Exhibit E-7
Risk Calculations for Existing Homes Using
Distribution for All Homes that Should Test
at Action Level 2pCi/L

Option A

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	TP	0.0428	2	19	175	0
3-4	TP	0.0293	2	62	390	True Positive
4-6	TP	0.0330	2	127	902	
6-8	TP	0.0116	2	212	527	
8-10	TP	0.0066	2	300	425	
10-20	TP	0.0062	2	470	624	3,196
>20	TP	0.0006	2	1,177	152	False Negative
2-3	FN	0.0289	NONE	19	(118)	
3-4	FN	0.0089	NONE	62	(119)	
4-6	FN	0.0043	NONE	127	(119)	
6-8	FN	0.0005	NONE	212	(24)	
8-10	FN	0.0001	NONE	300	(8)	
10-20	FN	0.0000	NONE	470	(4)	(392)
>20	FN	0.0000	NONE	1,177	(0)	

Option B

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	TP	0.0367	2	19	150	0
3-4	TP	0.0275	2	62	365	True Positive
4-6	TP	0.0322	2	127	881	
6-8	TP	0.0115	2	212	524	
8-10	TP	0.0066	2	300	424	
10-20	TP	0.0062	2	470	624	3,121
>20	TP	0.0006	2	1,177	152	False Negative
2-3	FN	0.0350	NONE	19	(143)	
3-4	FN	0.0107	NONE	62	(143)	
4-6	FN	0.0051	NONE	127	(140)	
6-8	FN	0.0006	NONE	212	(28)	
8-10	FN	0.0001	NONE	300	(9)	
10-20	FN	0.0000	NONE	470	(4)	(467)
>20	FN	0.0000	NONE	1,177	(0)	

Exhibit E-7 (continued)

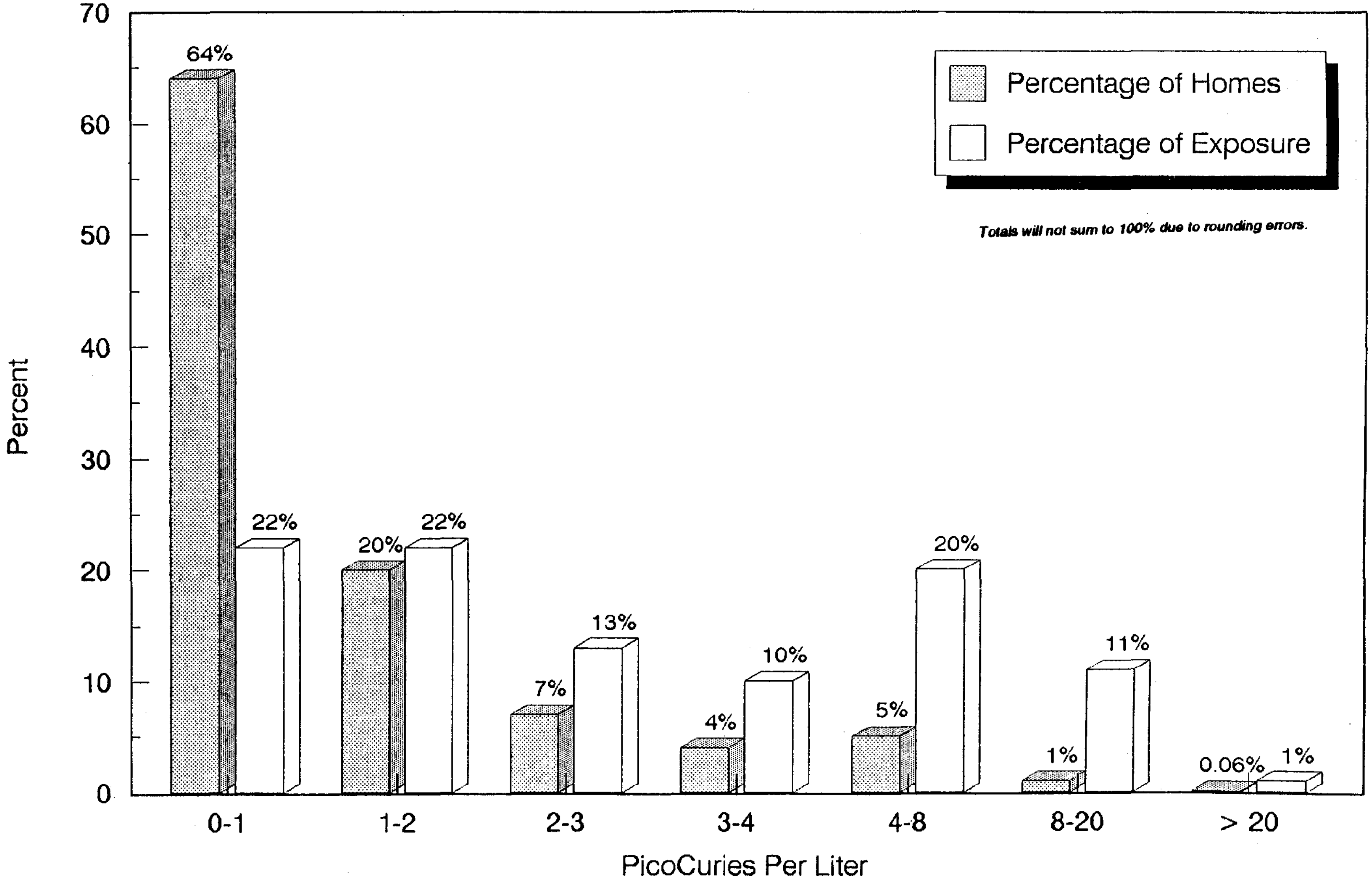
Option C

Radon pCi/l Interval	Test Status*	Fraction of Population	Radon Reduction Level	Annual Deaths Averted per Million Persons	Annual Lives Saved (Lost)	
0-1	FP/TN	0.6198	NONE	NONE	0	False Positive 0
1-2	FP/TN	0.2074	NONE	NONE	0	
2-3	TP	0.0372	2	19	152	
3-4	TP	0.0276	2	62	368	True Positive 3,127
4-6	TP	0.0323	2	127	883	
6-8	TP	0.0115	2	212	524	
8-10	TP	0.0066	2	300	424	
10-20	TP	0.0062	2	470	624	
>20	TP	0.0006	2	1,177	152	False Negative (460)
2-3	FN	0.0345	NONE	19	(141)	
3-4	FN	0.0106	NONE	62	(141)	
4-6	FN	0.0050	NONE	127	(138)	
6-8	FN	0.0006	NONE	212	(27)	
8-10	FN	0.0001	NONE	300	(9)	
10-20	FN	0.0000	NONE	470	(4)	
>20	FN	0.0000	NONE	1,177	(0)	

*True Positive (TP), False Positive (FP), True Negative (TN), and False Negative (FN). Assumes 100% testing and mitigation. Column entries may not sum due to rounding of estimates.

**There is no radon reduction level for TN homes, there is a radon reduction level of 2 for homes that test FP.

DISTRIBUTION OF HOMES AND TOTAL EXPOSURE AT SELECTED RADON LEVELS FOR ALL HOMES



APPENDIX F

RADON MITIGATION COST MODEL

EPA has a cost model for estimating the total system costs for radon mitigation in single-family detached homes. For any specified action level, the model can predict the total costs of reducing radon levels for homes above that level down to a specified "target level," an average level all homes will reach when mitigation systems are installed. The model calculates a weighted average cost (WAC) of mitigation, which can be multiplied by the number of housing units that EPA estimates will require radon mitigation to arrive at total mitigation costs. The costs are for contractor-installed systems (versus homeowner efforts). The estimates are provided as present value estimates--all unit costs are summed and presented in 1991 dollars based on a 3 percent discount rate.

Background

When the model was first built in 1987, the overall average target level of radon reduction was 4 pCi/L (equal to EPA's action level). For ease of development, the model's unit costs were developed using 4 pCi/L as a reference level for overall reduction. Scalers were devised to multiply against the reference unit costs for achieving a reduction down to 4 pCi/L to consider higher, or lower mitigation target levels. For instance, in some cases installing a mitigation system that would get homes above 4 pCi/L down to 2 pCi/L was estimated to be twice as expensive as a system that gets homes above 4 pCi/L down to 4 pCi/L. The model has been updated over time as EPA has learned more about mitigation experiences through the U.S., but it has kept this reference cost and scaler (multiplier) approach to estimating individual unit costs of various mitigation activities.

Recent Model Update

The model was updated in the Fall of 1991. EPA reviewed recent radon mitigation literature and consulted mitigation experts throughout the United States during its revision of the model. EPA's recently completed *Parametric Analysis of the Installation and Operating Costs of Active Soil Depressurization Systems for Residential Radon Mitigation* (EPA, October 1991) was relied on quite heavily for updating the costs of active subslab depressurization, which is the technology that EPA's expects the vast majority of homeowners to use. To a lesser degree, EPA's *Private Sector Radon Mitigator Survey* (June 1991) which reports on data collected in 1989 was also used.

Radon mitigation has been an active business for less than 10 years. There is not extensive published information on the usage rates of different systems and cost differences geographically. To overcome this lack of data, EPA consulted with expert radon mitigators throughout the country to develop information on what types of systems were being used and the costs of their usage in different parts of the country . These experts had significant experience and were mitigation instructors at EPA's Regional Radon Training Centers. They had substantial knowledge of practices in the Northeast, South, West, Pacific Northwest, and Central U.S. Additionally, a senior EPA mitigation expert, who has directed its extensive mitigation research activities for existing homes, and the principal developer of EPA's radon mitigation training course, who has substantial national experience in reducing radon levels in homes, were also consulted. From these experts' input and recognition of where most mitigation activity in the country would be occurring[1], cost functions and estimates of technology usage throughout the nation were prepared.

Model Components

The model has two major components. The first combines unit costs of different aspects of radon mitigation to derive total upfront and lifetime system costs for selected mitigation technologies that are most prevalently used in the U.S. The second component uses estimates of how often various technologies are used to provide weighted average costs of radon mitigation for upfront and lifetime expenditures. Each of these components is explained below.

Unit and Total Costs of Mitigation Systems

The model examines unit costs in five major cost categories: (1) diagnosis, (2) installation, (3) post-mitigation, (4) regular operation, and (5) repairs. It is supplied unit costs at a level of detail that is sensible to consider cost variations for four major factors: (1) foundation type, (2) mitigation technology, (3) initial radon levels, and (4) degree of difficulty in installing a system (i.e., whether homes are easy or hard to fix). For installation, it makes sense to provide unit costs for each combination of these factors. Therefore, average unit costs for all appropriate situations were developed (see Exhibit F-2). For diagnosis and post-mitigation, the best way to examine unit costs was on the basis of whether the home was easy or hard to fix (see Exhibit F-3). For regular operation costs (shown in Exhibit F-4) and repairs (shown in Exhibit F-5), the best way to examine unit costs was on the basis of the mitigation technology used.

Each of the cost categories used in the analysis is discussed below.

- Diagnosis - This category covers the average costs for a radon mitigator to provide a limited assessment of the extent of a radon problem in a home and to estimate the cost to fix it -- it is essentially the cost of a price estimate. It is delineated by whether the home will be easy or difficult to fix (see Exhibit F-3).

- Installation - This category covers the costs of materials and labor for initially installing the system (see Exhibit F-2). Unit costs were estimated for every mitigation situation (i.e., combination of factors) that the Agency believed should occur at levels that are not insignificant. It was assumed that any practice that would be used less than 5 percent of the time was insignificant. Note that the unit costs for homes between 2 and 4 pCi/L only apply if the action level considered in the cost analysis is either 2 or 3 pCi/L. For all technologies, it is assumed some sealing occurs in areas of the foundation where it makes sense to do so. (Sealing/Plugging as a mitigation technology category here means that its the only method used.)

- Post-mitigation - This category covers the contractor returning to the home shortly after system installation to check its operation and make any necessary adjustments. It is delineated by whether the home was easy or difficult to fix (see Exhibit F-3).

[1]Exhibit F-1 shows by EPA Region the percentage of homes that are above EPA's action level (4 pCi/L) based on EPA's report on the National Residential Radon Survey.

- Regular Operation - This category estimates the normal operating expenses incurred by homeowners during mitigation system operation (see Exhibit F-4). For all technologies, it is assumed to be at least $35 every 2 years (or about an average of $18 a year) based on the assumption that all homes will biennially test their radon levels (as recommended in the *RCP Interim Radon Mitigation Standards, 1991*) using an alpha track detector. Additional annual operating costs for each technology include:

  -- Active subslab depressurization - there are costs for fan electricity and heating/cooling losses of conditioned air.

  -- Simple ventilation - for this cost category, there are averaged costs of using ventilation fans with open windows and simply opening windows. The costs are for fan electricity (where a fan is used) and heating/cooling losses.

  -- Heat recovery ventilation - this technology has added costs for fan electricity, heat loss, annual filter replacement, and annual unit inspection.

  -- Depressurization in crawl space - there are costs for fan electricity and the added space heating requirements resulting from this system.

- Repairs - This cost category gives the costs of labor and materials for repairing mitigation systems, or replacing materials over time (see Exhibit F-5). The repair costs vary by mitigation technology:

  -- Active subslab depressurization - For all foundations, it is assumed that cracks will be resealed in years 2 and 20 after system installation to fix problems caused by caulks degrading or house settling. This resealing is estimated to cost $100 each time. Fans are replaced every 10 years at a cost of $150 each time and the alarm device that indicates fan failure is replaced for $95 in year 37 (half way through system life.) For crawl space homes, the membrane liner is replaced in year 37 (halfway through system life.)

  -- Sealing/plugging - initial sealing is repaired every five years at 40 percent of the original cost of installation.

  -- Simple ventilation - repair and replacement is averaged between using fans with open windows and open windows only. For all foundations, assumes cracks will be resealed in years 2 and 20. Basement and slab-on-grade homes that use fans replace them every 20 years for $230 each time. There are no replacement costs associated with open windows other than sealing foundation cracks.

  -- Heat recovery ventilation - costs are twice the ASD costs above, since the systems use two fans.

The unit costs are meant to reflect national average unit costs. This is after compiling the unit cost estimates mitigators provided from different regions of the country and considering where in the country certain mitigation practices in certain types of houses (by foundation) were likely to occur. As mentioned earlier, there was also consideration of areas of the country that will have the largest percentages of homes to fix.

In each case, the "average" size house that was considered had 1,900 square feet. Only major mitigation technologies that would often be used were considered -- active subslab depressurization (ASD), sealing/plugging, simple ventilation (using fans with open windows or just properly opened windows), heat recovery ventilation (HRV), and space depressurization for crawl space homes. Whether a house was easy or hard to fix related to its structural design, materials the home foundation rested on, and interior space use. The definition reflected the level of effort (and materials to some degree) a mitigator would spend on installing a system. Effort was always linked to whether areas in the house that required mitigator attention were readily accessible or not. For instance, whether cracks in the foundation that needed sealing were easy to reach or not. Often it would also depend on how much effort and materials were needed to install a system that would be fully effective in a particular situation. For example, where the area under a foundation does not provide for easy passage of soil gas, the resulting "poor communication" will lead to the placement of more than one suction pipe in an active subslab depressurization system. A house such as this would be "hard" to fix.

Initial radon levels, in pCi/L, were selected based on radon ranges that could lead to different selections of various types of mitigation systems and different overall levels of effort (although to a lesser degree). It is important to recognize that although mitigators are fixing homes with levels above 4 pCi/L, it is uncommon for homes initially testing between 2 to 4 pCi/L to be mitigated. Therefore, the expenses for mitigating homes in this range are based on theoretical assumptions of what the costs should be and not practical experience.[2] The housing foundation types selected represent the three dominant types that exist (note that partial and full basement homes are simply considered as basement homes). It was assumed that 47 percent of the homes have basements, 26 percent have slab-on-grade, and 27 percent have crawl space. These estimates were taken from the American Housing Survey for 1989 published by the Census Bureau.

The costs are estimated from the vantage point of the homeowner. For diagnosis, installation, post-mitigation, and repairs, the costs are the amounts homeowners would pay commercial contractors to perform the work. For the operating expenses, the costs reflect what homeowners would pay testing companies and utilities for electricity and space conditioning. It is important to consider that the first three costs elements together (diagnosis, installation, and post-mitigation) should sum to the total price mitigators would charge for mitigation system installation. It is assumed that mitigators are using practices EPA recommends for installing mitigation systems in its guidance materials and training programs. It is also important to recognize that the costs in each category are meant to provide average unit costs that can be combined to provide overall system costs. The costs considered in the analysis are for 74 years, the assumed average lifetime of a house.

The total costs for upfront expenses and system life are calculated for each type of mitigation system. The estimates are the summation of the appropriate unit cost components as shown in

[2]The expert mitigators were queried on whether it was reasonable to assume that the costs of getting the average house between 2 and 4 pCi/L down to 2 pC/l was the same as getting the average house between 4 and 8 pCi/L down to 4 pCi/L. They all believed such an assumption was reasonable.

Exhibits F-2 through F-5. Exhibit F-7 shows the total lifetime costs. An example of how these unit costs are combined is provided below:

> For basement homes that are greater than 20 pCi/L and easy to fix, their average cost is the summation of diagnosis ($80), installation ($1,000), post-mitigation ($70), annual operation ($110 per year, or $3,265 in present value terms), and repairs ($565 in present value terms), or $4,971. The initial costs that a mitigator would want to charge someone, $1,150, are captured by diagnosis, installation, and post-mitigation.

Mitigation Technology Usage

For each set of homes that is expected to exist in various pCi/L intervals, the model uses estimates of the percentage of time various technologies would be used in easy and hard situations. Exhibit F-6 shows the assumptions made by each type of home foundation. These percentages were arrived at by using the judgments of the expert mitigators mentioned above.

The left-hand portions of Exhibit F-7 show the cost results of the total lifecycle analysis in present value dollars for the case of all homes with an annual average level above 4 pCi/L reducing their radon levels to an overall annual average of 2 pCi/L. These initial results are used to develop a weighted average cost (WAC) for each foundation type and ultimately a WAC for the whole housing stock. This is calculated using information on the percentages of homes that are found in each range of initial radon levels and the percentages of homes with each foundation type.

Exhibit F-8 shows for each action level examined in Chapter 5 what percentage of homes were in different initial radon ranges. (EPA assumed an even distribution of range levels by foundation type.) In the cases where the action level is 4 pCi/L, the system costs for 2-4 pCi/L homes would not enter the analysis. The right-hand portion of Exhibits F-6 and F-7 show the case where the action level could be 2 or 3 pCi/L and homes between 2 to 4 pCi/L reduce their overall annual average to 2 pCi/L. As Exhibit F-8 shows, each of the action levels that was examined has different WACs for the upfront and total lifetime costs due to the variability in percentages of homes that are fixed at different levels. The lower WACs for lower action levels are due to an increasingly larger percentage of homes at lower radon levels that enter into the calculation of the WAC. In the majority of situations (especially in homes using ASD), the costs of reducing radon levels down to 2 pCi/L is about the same. However, mitigations in the smaller set of homes that use sealing and ventilation technologies cost less at the lower action levels because the mitigation systems need to provide lower percentage reductions in radon levels.

EXHIBIT F-1
ESTIMATED NUMBER OF HOUSING UNITS WITH RADON LEVELS ABOVE 4pCi/L, 1989–1990

REGIONS	SINGLE-FAMILY HOMES ABOVE 4pCi/L*	% OF TOTAL	MOBILE HOMES ABOVE 4pCi/L*	% OF TOTAL	MULTI-UNITS ABOVE 4pCi/L*	% OF TOTAL	TOTAL UNITS ABOVE 4pCi/L*	% OF TOTAL
REGION 1	124	3%	8	2%	80	5%	212	4%
REGION 2	189	5%	14	3%	179	12%	382	7%
REGION 3	526	14%	51	11%	173	12%	750	13%
REGION 4	555	15%	120	26%	192	13%	867	15%
REGION 5	1,122	29%	99	22%	424	29%	1,644	29%
REGION 6	205	5%	32	7%	66	5%	303	5%
REGION 7	596	16%	60	13%	158	11%	813	14%
REGION 8	407	11%	57	13%	141	10%	604	11%
REGION 9	56	1%	7	2%	29	2%	92	2%
REGION 10	33	1%	6	1%	11	1%	50	1%
TOTAL	3,812		453		1,453		5,719	

*Number of homes in thousands.
Source: National Residential Radon Survey

EXHIBIT F-2
RADON MITIGATION SYSTEM INSTALLATION COSTS BY BUILDING FOUNDATION, SOURCE STRENGTH, AND DEGREE OF DIFFICULTY TO FIX

		REDUCING HOMES ABOVE 4 pCi/L TO 4 pCi/L			REDUCING HOMES BETWEEN 2-4 pCi/L TO 2 pCi/L
SOURCE STRENGTH (pCi/L):		>20	8-20	4-8	2-4
	DIFFICULTY TO FIX				
BASEMENT					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	$1,000	$1,000	$1,000	$1,000
	HARD	1,500	1,300	1,300	1,300
SEALING/PLUGGING	EASY	1,250	1,100	500	500
	HARD	0	3,000	2,750	2,750
SIMPLE VENTILATION*		0	0	115	115
HEAT RECOVERY VENTILATION*		0	2,000	2,000	0
SLAB-0N-GRADE					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	$1,200	$1,200	$1,200	$1,200
	HARD	1,700	1,600	1,600	1,600
SEALING/PLUGGING	EASY	0	600	600	600
	HARD	0	3,000	1,250	1,250
SIMPLE VENTILATION*		0	0	115	115
HEAT RECOVERY VENTILATION*		0	0	0	0
CRAWLSPACE					
ACTIVE SUBMEMBRANE DEPRESSURIZATION	EASY	$1,300	$1,300	$1,300	$1,300
	HARD	1,650	1,650	1,650	1,650
ISOLATION	EASY	0	1,250	500	500
	HARD	0	0	1,250	1,250
SIMPLE VENTILATION*		0	0	0	400
DEPRESSURIZATION*		450	450	450	450

MULTIPLIERS FOR HOMES ABOVE 4 pCi/L ACHIEVING 2PCi/L LEVELS (MULTIPLIED AGAINST COSTS FOR REACHING 4 pCi/L)

ACT.SUBSL. DEPPRESURIZATION - EASY:	*1.00*
ACT.SUBSL. DEPPRESURIZATION - HARD:	*1.25*
SEALING/PLUGGING - EASY AND HARD:	*1.50*
SIMPLE VENTILATION (BASEMENT AND S-O-G):	*2.00*
SIMPLE VENTILATION (CRAWLSPACE):	*1.00*
HEAT RECOVERY VENTILATION:	*1.75*
DEPRESSURIZATION:	*1.00*

HOMES BETWEEN 2-4 pCi/L REACH 2 WITH SAME ASSUMED COST AS HOMES BETWEEN 4-8 pCi/L REACHING 4 pCi/L

* Simple ventilation and depressurization (in the crawlspace) are assumed to be easy and heat recovery ventilation is assumed to be hard to do.

Note: Zero entries in the installation table indicates that the technology shown would not be applied in that particular situation.

EXHIBIT F-3
COMMON COSTS ASSOCIATED WITH PROPER MITIGATION SYSTEM INSTALLATION FOR ALL FOUNDATIONS BY DEGREE OF DIFFICULTY TO FIX*

	FOR HOMES REDUCED TO 4 PCi/L		MULTIPLIERS FOR ACHIEVING 2 PCi/L FOR HOMES INITIALLY ABOVE 4 pCi/L**
	EASY	HARD	
DIAGNOSTICS (PRICE ESTIMATE)	$80	$120	1.0 - Same cost for homes going down to 2 pCi/L
POSTMITIGATION	$70	$280	Multiply by 1.0 for easy homes and 1.5 for hard homes going down to 2 pCi/L

*Note that simple ventilation and crawlspace depressurization is always considered easy to install and heat recovery is always considered hard to install.

**Homes between 2 and 4 pCi/L that reduce to 2 pCi/L are assumed to have the same costs as homes above 4 pCi/L reducing down to 4 pCi/L (i.e., the multiplier is 1.0).

EXHIBIT F-4
ANNUAL OPERATING COSTS FOR RADON MITIGATION SYSTEMS

	All Foundations	
SEALING/PLUGGING COSTS (FOR BIENNIAL TESTING) FOR HOMES TO REACH 4 OR 2 pCi/L	$35 (or $18 per year)	
FOR HOMES INITIALLY ABOVE 4 pCi/L		
	Basement & Slab-on-grade	Crawlspace
ACTIVE SUBSLAB DEPRESSURIZATION		
FOR HOMES AT 4 PCi/L:	$110	$110
FOR HOMES AT 2 PCi/L:	110	110
(Includes annual testing, energy penalty, and fan electricity)		
SIMPLE VENTILATION		
FOR HOMES AT 4 PCi/L:	$335	$120
FOR HOMES AT 2 PCi/L:	670	120
(Includes annual testing, energy penalty, and fan electricity)		
DEPRESSURIZATION		
FOR HOMES AT 4 PCi/L:	-	$185
FOR HOMES AT 2 PCi/L:	-	185
(Includes annual testing, energy penalty, and fan electricity)		
HEAT RECOVERY VENTILATION		
FOR HOMES AT 4 PCi/L:	$240	-
FOR HOMES AT 2 PCi/L:	480	-
(Includes annual testing, energy penalty, and fan electricity)		
FOR HOMES BETWEEN 2 AND 4 pCi/L REACHING TARGET LEVEL OF 2 pCi/L		
Assumed that costs are the same as homes above 4 pCi/L reaching 4 pCi/L target level (i.e., multiplier of 1.0)		

Note: Operating costs are assumed constant across homes with different source strengths (i.e., initial radon levels) and degree of difficulty to fix.

EXHIBIT F-5
REPAIR COSTS FOR COMPONENTS OF RADON MITIGATION SYSTEMS REDUCING HOMES ABOVE 4 pCi/L DOWN TO ALTERNATIVE TARGET LEVELS

FOR HOMES ACHIEVING 4 pCi/L

ACTIVE SUBSLAB DEPRESSURIZATION:	REPLACING FANS EVERY 10 YEARS AT A COST OF $150 EACH TIME WARNING DEVICE FAILURE AT YEAR 37: $95
SUBMEMBRANE DEPRESSURIZATION IN CRAWLSPACE:	1/2 INSTALLATION (EASY) AT YEAR 37 WARNING DEVICE FAILURE AT YEAR 37: $95
SEALING:	RESEALING EVERY 5 YEARS AT 40% OF INSTALLATION COST
SIMPLE VENTILATION:	REPLACING FANS EVERY 10 YEARS FOR $230
HEAT RECOVERY VENTILATION:	DOUBLING THE COST OF ASD ABOVE (SYSTEMS USE TWO FANS)

FOR HOMES ACHIEVING 2 pCi/L

REPLACEMENT COST MULTIPLIERS FOR HOMES ACHIEVING 2 pCi/L LEVELS
(MULTIPLY THEM AGAINST ABOVE COSTS OF GETTING HOMES DOWN TO A TARGET LEVEL OF 4 pCi/l)

ACTIVE SUBSLAB DEPRESSURIZATION - EASY:	1.00
ACTIVE SUBSLAB DEPRESSURIATION - HARD:	1.25
SEALING/PLUGGING:	1.50
SIMPLE VENTILATION (NON-CRAWLSPACE):	2.00
HEAT RECOVERY VENTILATION:	1.75
CRAWLSPACE DEPRESSURIZATION:	1.00
CRAWLSPACE SIMPLE VENTILATION (Only 2-4 Range):	1.00

FOR HOMES BETWEEN 2 TO 4 pCi/L ACHIEVING 2 pCi/L

COSTS ARE THE SAME AS HOMES FROM 4 TO 8 pCi/L REACHING 4 pCi/L
(i.e., the multiplier is 1.0)

Note: Replacement costs are assumed constant across homes with different source strengths (i.e., initial radon levels) and degrees of difficulty to fix.

EXHIBIT F-6
MITIGATION TECHNOLOGY USE BY BUILDING FOUNDATION TYPE, SOURCE STRENGTH, AND DEGREE OF DIFFICULTY TO FIX

SOURCE STRENGTH pCi/L:		>20	8-20	4-8	2-4
	DIFFICULTY				
BASEMENT					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	65%	65%	50%	45%
	HARD	35%	30%	20%	15%
SEALING/PLUGGING	EASY	0%	0%	10%	15%
	HARD	0%	0%	5%	5%
SIMPLE VENTILATION	(EASY)	0%	0%	10%	20%
HEAT RECOVERY VENTILATION	(HARD)	0%	5%	5%	0%
SLAB-ON-GRADE					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	65%	65%	60%	60%
	HARD	35%	35%	30%	30%
SEALING/PLUGGING	EASY	0%	0%	5%	5%
	HARD	0%	0%	0%	0%
SIMPLE VENTILATION	(EASY)	0%	0%	5%	5%
HEAT RECOVERY VENTILATION	(HARD)	0%	0%	0%	0%
CRAWLSPACE					
ACTIVE SUBMEMBRANE DEPRESSURIZATION	EASY	55%	55%	55%	35%
	HARD	35%	35%	30%	15%
ISOLATION	EASY	0%	0%	5%	5%
	HARD	0%	0%	0%	10%
DEPRESSURIZATION	(EASY)	10%	10%	10%	10%
SIMPLE VENTILATION	(EASY)	0%	0%	0%	25%

Note: Technologies with less than 2.5% usage were given 0%; all %'s rounded to nearest 5%.

EXHIBIT F-7
SAMPLE MODEL OUTPUT
RADON MITIGATION SYSTEM COSTS BY PICOCURIE/LITER RANGE
(PRESENT VALUE 1991$)

			REDUCING HOMES TO 2 pCi/L		
SOURCE STRENGTH (pCi/L):		>20	8-20	4-8	2-4
	DIFFICULTY TO FIX				
BASEMENT					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	$4,971	$4,971	$4,971	$4,971
	HARD	6,377	6,127	6,127	5,521
SEALING/PLUGGING	EASY	0	0	3,078	2,280
	HARD	0	0	14,249	9,717
SIMPLE VENTILATION		0	0	21,091	10,620
HEAT RECOVERY VENTILATION		0	19,840	19,840	0
SLAB-ON-GRADE					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	$5,171	$5,171	$5,171	$5,171
	HARD	6,627	6,502	6,502	5,821
SEALING/PLUGGING	EASY	0	0	3,557	2,599
	HARD	0	0	0	0
SIMPLE VENTILATION		0	0	21,091	10,620
HEAT RECOVERY VENTILATION		0	0	0	0
CRAWLSPACE					
ACTIVE SUBSLAB DEPRESSURIZATION	EASY	$5,509	$5,509	$5,509	$5,509
	HARD	6,862	6,862	6,862	6,109
ISOLATION	EASY	0	0	3,078	2,280
	HARD	0	0	0	4,925
DEPRESSURIZATION		6,640	6,640	6,640	6,640
SIMPLE VENTILATION		0	0	0	4,539

EXHIBIT F-8
DISTRIBUTION OF HOMES INITIALLY TESTING ABOVE SELECTED ACTION LEVELS AND WEIGHTED AVERAGE COSTS OF MITIGATION

	DISTRIBUTION OF HOMES ABOVE THE ACTION LEVEL IN pCi/L*					WEIGHTED AVERAGE COST (1991$)	
ACTION LEVEL	>20	8-20	4-8	2-4	TOTAL	UPFRONT	LIFETIME
2 pCi/L	0.0156	0.1258	0.3349	0.5237	1.00	$1,366	$6,126
3 pCi/L	0.0257	0.2073	0.5047	0.2623	1.00	1,442	6,359
4 pCi/L	0.0390	0.3119	0.6491	0.0000	1.00	1,520	6,566

*Rows may not sum due to rounding.

APPENDIX G

SENSITIVITY ANALYSIS OF THE COST-EFFECTIVENESS OF THE REVISED *CITIZEN'S GUIDE*

This appendix presents the results of a sensitivity analysis of the most significant parameters used in the cost-effectiveness analysis in Chapter 5. It only examines the option that EPA has decided to adopt in the revised *Citizen's Guide*, which recommends that the public fix all homes with radon levels above 4 pCi/L. Exhibit G-1 shows the results of the analysis for this option or **Base Case**. To facilitate evaluation of the sensitivity of the cost-effectiveness results to the assumptions that were used for the most critical parameters, the base case results appear at the top of the exhibit.

Each major parameter that EPA used was analyzed independently with regard to the relevant range of values it potentially could have. All the other values remained as they were in the base case (i.e., were held constant) while each parameter was examined. In the base case, the critical assumptions were:

- radon's risk factor is equal to EPA's central estimate;
- 100 percent of the public will follow EPA's testing and mitigation advice;
- 91 percent of the public will conduct a short-term follow-up test, and the rest will conduct a long-term follow-up test;
- the time period of analysis is 74 years;
- on average, all homes installing radon mitigation systems will reduce radon levels to 2 pCi/L;
- the radon testing policy covers all single family homes, multi-unit and group quarters below the third floor, and mobile homes on permanent foundations;
- the social discount rate[1] is 3 percent;
- mitigation systems have an average present value cost of $6,566 (about 25 percent of the total expenses are initial costs and about 75 percent of the costs are operating and maintenance (O&M) costs); and
- smoking habits will continue unchanged.

The following assumptions were analyzed in the sensitivity analysis.

- <u>Risk factor</u> - Used EPA's lower and upper bound estimates of the radon risk factor. (See Chapter 2 for values used.)

[1]The opportunity cost of money used by homeowners to test, fix, and operate their homes.

- Public Response rates - Examined changes in the expected level of radon testing and the expected level of mitigation that would occur after testing.

- Testing choices - Analyzed the public's use of a long-term follow-up test (Option A) in all cases or use of a short-term follow-up test (Option B) in all cases. EPA recommends in the 1992 *Guide* that the public use either test.

- Time period of analysis - Examined shorter time periods for the analysis that reflect other relevant time periods EPA could have considered. Five years represents the lowest available estimate of how often homes are sold on average. Thirty years is the life of a typical mortgage.

- Effectiveness of systems - Selected a level of 3 pCi/L and 1 pCi/L as alternatives to EPA's assumption that radon mitigation will attain an annual average reduction to 2 pCi/L. Alternatively, it was assumed that homes between 1 and 2 pCi/L that have false positive results could get down to 1 pCi/L and homes above 2 pCi/L would get down to that level. In a subsequent run, it was assumed that homes with false positives between 1 and 2 pCi/L could get down to 1 pCi/L and homes between 0 and 1 pCi/L could reduce their radon levels by 50 percent if mitigation systems are installed while homes above 2 pCi/L are still reduced down to 2 pCi/L.

- Coverage of testing policy - Excluded from consideration all housing units that were not single family homes. Single family homes were assumed to average 2.8 residents per unit as opposed to 2 residents per home in other types of housing units.

- Discount rate - Examined other discount rates that have been used by the federal government in preparing cost analyses.

- Initial mitigation system costs - Varied by 50 percent the upfront costs of installing radon mitigation systems (including the costs of diagnosis, installation, and post mitigation follow-up).

- Mitigation O&M cost - Varied by 30 percent the costs of operating and repairing radon mitigation systems over their entire lives.

- Smoking Habit Changes - Analyzed two types of changes in smoking patterns. First, EPA examined changes in habits that reduced the level of current smokers (by 20 to 50 percent) and assumed that this share of people entered the "former smoker" category. Second, EPA analyzed changes in smoking habits that altered the distribution of "current smokers" and "never smokers." In this case, assumed that 20 to 50 percent of today's current smokers are instead persons who have never smoked.

EXHIBIT G-1

SENSITIVITY ANALYSIS OF THE COST-EFFECTIVENESS OF THE REVISED CITIZEN'S GUIDE

Case	Annual Lives Saved	Annualized Cost (Million 1991$)	Cost Per Life Saved (Thousand 1991$)
BASE CASE	2,230	$ 1,500	$ 670
RISK FACTOR			
EPA Lower Bound Risk Factor	1,400	1,500	1,070
EPA Upper Bound Risk Factor	5,900	1,500	260
PUBLIC RESPONSE RATE			
50% Testing/100% Mitigation	1,120	750	670
10% Testing/100% Mitigation	220	150	670
100% Testing/25% Mitigation	560	440	800
100% Testing/10% Mitigation	220	230	1,040
TESTING CHOICES			
100% Option A	2,160	930	430
100% Option B	2,240	1,560	700
TIME PERIOD OF ANALYSIS			
5 Years	2,230	3,930	1,760
30 Years	2,230	1,730	780
EFFECTIVENESS OF SYSTEMS			
Average Reduction to 3 pCi/L	1,710	1,500	880
Average Reduction to 1 pCi/L	3,000	1,500	490
Homes Between 1 and 2 Can Reduce to 1 pCi/L	2,270	1,500	660
Homes Between 1 and 2 Can Reduce to 1 pCi/L and Homes Below 1 Can Reduce by 50%	2,280	1,500	660
COVERAGE OF TESTING POLICY			
Single-Family Homes Only	2,120	1,240	590
ASSUMED DISCOUNT RATE			
0% Discount Rate	2,230	1,240	560
5% Discount Rate	2,230	1,730	780
7% Discount Rate	2,230	1,980	890
10% Discount Rate	2,230	2,360	1,060
INITIAL MITIGATION SYSTEM COSTS			
50% Cost Reduction	2,230	1,340	600
50% Cost Increase	2,230	1,670	750
MITIGATION SYSTEM O&M COSTS			
30% Cost Reduction	2,230	1,180	530
30% Cost Increase	2,230	1,830	820
DIFFERENCES IN SMOKING HABITS			
20% Current Smokers Quit	2,060	1,500	730
50% Current Smokers Quit	1,790	1,500	840
20% of Current Smokers Never Smoked	1,930	1,500	780
50% of Current Smokers Never Smoked	1,490	1,500	1,010

APPENDIX H

COST-EFFECTIVENESS OF RADON ACTION LEVELS GREATER THAN 4 pCi/L

EPA initially examined five different action levels for the revised *Citizen's Guide*. It quickly became apparent that the higher action levels did not reduce nearly as much of the risk from radon as did the lower action levels. Also, the lower action levels were found to be incrementally cost-effective. Reducing radon levels in the additional homes covered by lower action levels appeared to be a good risk reduction purchase for the public to make if practical programs for doing so could be offered.

Exhibit H-1 provides a summary of the results of EPA's analysis of action levels greater than and lower than 4 pCi/L. The same approach that is explained in Chapter 5 and supporting appendices was used to provide these results. For the action levels of 8 and 20 pCi/L, EPA estimates that 2.1 million homes and 0.2 million homes, respectively, would be mitigated if a 100 percent public response rate were obtained. This compares with 6.4 million homes that would be mitigated when the action level is 4 pCi/L.

EXHIBIT H-1
COST PER LIFE SAVED
UNDER ALTERNATIVE TESTING AND MITIGATION PROGRAMS

Action Level	Number of Lives Saved Annually	Annualized Cost (1000s of 1991$)	Average Cost per Life Saved (1000s of 1991$)	Incremental Cost per Life Saved (1000s of 1991$)
2 pCi/L	3,100	3,421,000	$1,100	$2,400
3 pCi/L	2,600	2,181,000	800	1,700
4 pCi/L	2,200	1,504,000	700	900
8 pCi/L	1,100	501,000	400	400
20 pCi/L	220	116,000	500	500[a/]

[a/] Based on assumption that "no action" was the alternative EPA had to this action level.

Note: The central estimate of the radon risk factor is used in this analysis, rather than the upper bound risk estimate. The upper bound estimate would have increased all the risk estimates by about 2.5 times and reduced the cost-effectiveness estimates by about 60 percent.

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