RAC Report No. 15-RSALOP-RFSAL-1999-FINAL

FINAL REPORT

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

October 1999

Submitted to the Radionuclide Soil Action Level Oversight Panel
in Partial Fulfillment of Contract between RAC and the Rocky Flats Citizen's Advisory Board

Risk Assessment Corporation

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EXECUTIVE SUMMARY

The Rocky Flats Environmental Technology Site (RFETS) is owned by the U.S. Department of Energy (DOE) and is currently operated by Kaiser-Hill Company. For most of its history, the Dow Chemical Company operated the Rocky Flats Plant as a nuclear weapons research, development, and production complex. The Rocky Flats Plant is located about 8–10 km (5–6 mi) from the cities of Arvada, Westminster, and Broomfield, Colorado and 26 km (16 mi) northwest of downtown Denver, Colorado. This current project is evaluating the radionuclide soil action levels developed for implementation by the DOE, the Environmental Protection Agency (EPA) and the Colorado Department of Public Health and Environment (CDPHE) (DOE/EPA/CDPHE 1996). Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which remedial action should be considered to prevent people from receiving radiation doses larger than a predesignated limit. As a result of public concern about the proposed soil action levels, DOE provided funds for the Radionuclide Soil Action Level Oversight Panel (RSALOP) to select a contractor to conduct an independent assessment and to calculate soil actions levels for the RFETS. *Risk Assessment Corporation* (*RAC*) was selected to carry out the study.

RAC is using several environmental assessment computer programs, in particular, the RESRAD computer program, to calculate the soil action levels for this project. The purpose of Task 3, *Inputs and Assumptions*, was to evaluate the importance of input parameters and assumptions used to calculate the dose and soil action levels for cleanup at the RFETS. The task involved performing a sensitivity analysis using RESRAD to identify those parameters that have the greatest impact on the outcome of the soil action level calculation. For the parameters that were important to the final outcome, the task required *RAC* to develop site-specific values if data were available or to create uncertainty distributions of values from published literature. The sensitivity analysis was a single-parameter analysis, where a range of values for one parameter at a time was evaluated. *RAC* used the latest version of the RESRAD code (Version 5.82) to carry out the sensitivity analysis. This version is an update from the version used in the previous soil action level assessment (DOE/EPA/CDPHE 1996). In general, the newer version is a windows-based application of earlier versions of RESRAD. There is, however, one major conceptual difference in the formulation of the resuspension pathway. This difference decreases the importance of inhalation in terms of the total dose. In light of this, *RAC* used site-specific data to simulate resuspension for Rocky Flats outside the RESRAD code.

Of over 50 parameters assessed for their influence on the final result, four parameters were found to have the greatest impact on the final results:

- Soil-water equilibrium distribution coefficient
- Area of contamination
- Mass loading factor
- Mean annual wind speed.

Most Sensitive Parameters

The majority of this report focuses on these four parameters and provides parameter values or uncertainty distributions for them based on site-specific data or on literature values. The probability distributions describe the uncertainty in the values that arises from natural variability or from lack of knowledge about a particular parameter. This concept and the development of the parameter values and/or distributions are described in detail in this report. The following table summarizes the differences in parameter values or method of evaluation between the previous DOE/EPA/CDPHE assessment and the *RAC* approach.

Table ES-1. Values for the Four Most Sensitive Parameters for the Independent Calculation and Comparison with those from the DOE/EPA/CDPHE Assessment

Parameter	DOE/EPA/CDPHE value	*RAC* value
Distribution coefficient	Deterministic Pu = 218 $cm^3\ g^{-1}$ Am = 76 $cm^3\ g^{-1}$ U = 50 $cm^3\ g^{-1}$	Treated stochastically based on Rocky Flats measurements and literature values; median values (GSD[a]) of Pu = 2300 $cm^3\ g^{-1}$ (5.6) Am = 1800 $cm^3\ g^{-1}$ (8.1) U = 2.3 $cm^3\ g^{-1}$ (5.4)
Area of contaminated zone	40,000 m^2	Defined based on historic soil concentration measurements at Rocky Flats (see report text)
Mass loading	0.000026 g m^{-3}	Model will be calibrated based on results of soil and airborne concentration (see report text)
Mean annual wind speed	Not required for RESRAD Version 5.61	Use a 5-year annual average STAR data set collected at Rocky Flats met station

[a]GSD = geometric standard deviation, which is a measure of the extent of the distribution

The distribution coefficient was important in the Radionuclide Soil Action Level assessment because it defines the relationship of the concentration of the contaminant in the soil to the concentration of the contaminant in water, and it can influence calculations involving contaminants in the groundwater. *RAC* included groundwater as a source of water in the rancher, child of rancher, and infant of rancher scenarios, so it was important to carefully consider all data in establishing a value or range of values for this parameter. The distribution coefficient, called the K_d value, can extend over a very wide range even for a single type of soil so *RAC* realized it was essential to incorporate as much data as possible in their assessment. We expanded the bounds of the distribution coefficients reported previously by creating a distribution of values for uranium, plutonium, and americium based on a further review of the literature and the use of site-specific data. In the *RAC* assessment, the distribution for each radionuclide was further defined by the geometric standard deviation, which gives an estimate of how much uncertainty there is about the midpoint of the distribution.

The area of contaminated zone is a parameter required in the RESRAD code that defines a specified area in which the contamination is uniformly distributed. Unfortunately, for much of the area around Rocky Flats, especially east of the 903 Area, the plutonium concentrations vary by more than 100 times. This made it difficult to assume a uniform area of contamination and still have a large enough area where contamination was defined. To address this issue, *RAC* compiled historic soil monitoring data from the Rocky Flats area to create contours of contamination at and surrounding the 903 Area. These data represent the actual contamination in soil and were used in RESRAD to calculate soil action levels.

The term mass loading was used in this analysis as a measure of resuspension of soil from the ground. Resuspension is a complex process that is affected by many environmental factors that have not been well quantified The previous DOE/EPA/CDPHE assessment used a value of 0.000026 g m^{-3} for mass loading factor to represent resuspension. The current version of RESRAD uses a mass loading factor to define resuspension, but even the developers of RESRAD stressed its inadequacy at representing actual conditions at a given site. As a result, *RAC* used historic air monitoring data collected at Rocky Flats as the best measure of resuspension. *RAC* considered the location of each scenario onsite where the hypothetical person resides and/or works, and used actual air monitoring data in combination with the site-specific soil contamination data described above to set up a relationship between concentrations in air and soil to estimate resuspension. This approach bypassed the area factor calculation in RESRAD and defined resuspension based on actual air monitoring data. A more extensive discussion of this approach is outlined in Task 5, *Independent Calculation.*

The mean annual wind speed was not required in the previous version of RESRAD, so the DOE/EPA/CDPHE assessment did not specify a value for this parameter. Because *RAC* estimated resuspension based on site-specific air monitoring data, it was important to also use site-specific meteorological data. *RAC* used 5-year average wind speed and atmospheric stability class information from the onsite Rocky Flats meteorological station. High wind events occur in the Rocky Flats area and were evaluated in the Historical Public Exposure Studies on Rocky Flats for their effect on moving contamination from the 903 Area before it was covered with an asphalt pad. High winds also result in lower air concentrations than would be expected if the same material was dispersed over a longer period of time during average wind speed conditions. As a result, high wind events were not evaluated further in this assessment.

Less Sensitive Parameters

Six parameters were found to affect the outcome of the calculation only slightly:

- Cover depth (depth of soil that must be removed to reveal the contaminated soil)
- Fraction of the total outside air contamination that is available indoors (indoor dust filtration)
- Soil-to-plant transfer factors
- Depth of soil mixing layer (depth of uniform contamination)
- Fraction of irrigation water contaminated by groundwater
- Thickness of contaminated zone (non-uniformly distributed).

For these somewhat sensitive parameters *RAC* used the values from the DOE/EPA/CDPHE assessment for cover depth and indoor dust filtration. For the other four, values more consistent with studies published in the open scientific literature were selected. For the depth of soil mixing layer, or the depth over which soil is uniformly distributed, *RAC* selected a value of 0.03 m, instead of 0.15 m, based on published studies at Rocky Flats. For the thickness of the contaminated zone, *RAC* selected a value of 0.20 m, instead of 0.15 m, based on studies that show the contamination is distributed over the top 20 cm (0.20 m) of soil with very little movement of the contamination over the past 20 years. For the fraction of irrigation water contaminated by groundwater (irrigation water contamination fraction), *RAC* determined that groundwater might be used for irrigation or as a source of drinking water. As a result, they assumed that all of the groundwater used for irrigation would be contaminated (irrigation contamination fraction = 1.0). In the previous assessment, it was assumed that none of the water would be contaminated (irrigation contamination fraction = 0).

Soil-to-plant transfer factors quantify that portion of contamination in soil that is transferred to plants via root uptake. The previous DOE/EPA/CDPHE assessment used a deterministic approach, while *RAC* treated these factors stochastically based on the recent National Council on Radiation Protection and Measurement Report No. 129, *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies* (NCRP 1999). This screening methodology suggested distributions for soil-to-plant transfer factor that reflect uncertainty resulting from different soil conditions, soil types, and soil chemistry.

Other Parameters

The other parameters required to run the RESRAD code were not sensitive to changes in values; therefore, *RAC* did not give additional effort to changing or revising the values from those used in the previous assessment. For some parameters, *RAC* changed the previous value somewhat, or the method of calculating the parameter value, based on a consistent approach. For example, *RAC* used an external gamma shielding factor of 0.7, along with the time spent indoors, outdoors, and offsite to calculate occupancy factor. This method was more straightforward than that used previously.

This report also summarizes current studies that clearly show that plutonium in the soil at Rocky Flats is insoluble and, thus, may not get into the groundwater. However, *RAC* has included the groundwater pathway in the rancher, child of rancher, and infant of rancher scenarios, and this report describes the approach used to study the sensitivity of the drinking water pathway when contaminated groundwater is assumed as the source. This assessment showed that groundwater can have an impact on dose that needs to be recognized. Because of the severe limitations on time and resources in this study, *RAC* recommended that a future study be directed toward this type of work, particularly looking at the migration of ^{241}Am and its progeny. Groundwater pathways are assessed in this project on a screening basis only.

Another important parameter for RESRAD is the initial concentrations of radionuclides. In the previous assessment, DOE/EPA/CDPHE defined the initial concentrations of each radionuclide of interest as 100 pCi g^{-1}. In contrast, *RAC* used the measured soil concentration data to determine actual soil concentrations, initialized to the year that the soil action level calculations began. The concentrations of ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, and ^{241}Am are given relative to $^{239+240}$Pu. This technique clarifies the RESRAD results for the user by building in the appropriate site-specific ratios of radionuclides in the calculation of action levels. Soil concentrations for uranium at Rocky Flats are primarily located in hot spots. In Task 5, *RAC* calculates a soil action level for uranium based on the concentration of uranium in hot spots, as determined from the available literature. This report also provides the most recent values for inhalation and ingestion dose conversion factors that will be used in the independent calculation in Task 5.

Scenarios

The Task 3 report describes the seven scenarios that are currently being evaluated: the three scenarios described in the previous assessment, *Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement*, dated October 31, 1996 (DOE/EPA/CDPHE 1996), along with four additional scenarios that *RAC* has proposed after numerous discussions with the RSALOP at the monthly soil action level meetings. Parameter values for the DOE/EPA/CDPHE (residential, open space user, and office worker) and *RAC* scenarios (rancher, child of rancher, infant of rancher, and current onsite industrial worker) are summarized in this report. In designing the scenarios, *RAC* carefully considered offsite exposures so that if the person living onsite full-time is protected, then the person living offsite will be protected. Selecting parameter values for breathing rate and soil ingestion are described in detail. Based on published breathing rate studies, *RAC* defined distributions of breathing rates for active and sedentary adults, children, and infants. Using these distributions and the recommended breakdowns of daily activity for each scenario, *RAC* created distributions of scenario breathing rates and selected the 95th percentile value from that distribution for the annual breathing volume. A similar process was used to establish soil ingestion rates for the hypothetical individuals in the scenarios. While soil ingestion rates based on studies conducted from a few days to a few weeks are valid and important, it is necessary to carefully consider the implications of translating this daily soil ingestion rate to an annual soil ingestion rate. For these reasons, *RAC* selected the 50th percentile, or median, of the distribution as the daily soil ingestion rate for the scenarios.

Some scenario-related parameter values were different from those in the previous assessment. Because *RAC* included the drinking water pathway in their assessment, they provided an annual drinking water intake of 730 L y^{-1} for the adult rancher, and appropriate values for the child and infant scenarios. The DOE/EPA/CDPHE scenarios did not include drinking water exposure as a potential pathway. *RAC* recommended higher annual consumption rates than those used in the DOE/EPA/CDPHE assessment for fruits, vegetables, and grains based on published literature values. *RAC* also recommended values for milk and meat consumption, exposure pathways not considered in the DOE/EPA/CDPHE assessment. All scenario-related parameters were treated deterministically in this analysis.

In conclusion, Task 3 was focused primarily on those parameters that influence the outcome of the soil action level calculation to the greatest extent. For RESRAD, the most sensitive parameters were mass loading, distribution coefficients, area of contamination, and mean annual wind speed. Important scenario-related parameters were the breathing rate and soil ingestion rates. These values and distributions of values presented in this report will be used to calculate soil action levels and dose reported in Task 5, *Independent Calculation*.

CONTENTS

FIGURES

TABLES

TASK 3: INPUTS AND ASSUMPTIONS

INTRODUCTION

Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which remedial action should be considered to prevent people from receiving radiation doses larger than a predesignated limit. The soil action levels for radionuclides calculated for the Rocky Flats Environmental Technology Site (RFETS) by the U.S. Department of Energy (DOE), U.S. Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE) are being reevaluated because of public concern and interest in the methods previously used and the recommended soil action levels proposed. A Radionuclide Soil Action Level Oversight Panel (RSALOP) was established and a contractor was hired to conduct an independent assessment and calculate soil action levels for the Rocky Flats site. *Risk Assessment Corporation (RAC)* was hired to perform the study. The Rocky Flats Citizen's Advisory Board is administering a grant provided by DOE for the review.

The primary goal of Task 3 was to report the results of a sensitivity analysis conducted on the inputs and assumptions required for using the RESRAD computer code. Site-specific values were derived or uncertainty distributions were created for critical parameters emerging from the sensitivity analysis. The sensitivity of each parameter was assessed using the built-in Monte Carlo-based sensitivity analysis packaged with the latest version of RESRAD. This sensitivity analysis package does not operate in a traditional Monte Carlo mode; rather, it allows the user to input a range of possible values for a parameter, and the endpoints of this range are evaluated separately to show the change in the output result for these different input values. Also included in the Task 3 report is the careful evaluation of scenarios for their applicability to potential future land uses. This report describes the process of scenario evaluation and reports the scenarios chosen for the independent analysis.

A Monte Carlo interface for RESRAD has been developed and tested by *RAC* for use in Task 5, *Independent Calculation*. This interface uses the distributions identified in this task to develop uncertainties for dose and soil action level for each of the scenarios. The Monte Carlo package developed by *RAC* uses the probability distributions given in this report as inputs for a stochastic calculation of dose and soil action levels. The interface is calibrated to reflect site-specific conditions and apply available site-specific historic data, particularly air monitoring and soil concentration data. Results of these independent calculations of dose and soil action level will be reported in Task 5.

Parameters Explored

Important parameters for which distributions and/or site-specific values were developed were identified by using a sensitivity analysis. The sensitivity analysis was a single-parameter analysis, where a range of values for one parameter at a time was explored to determine its impact on the final result. These ranges of values were explored using the built-in Monte Carlo-based tool in RESRAD Version 5.82. If the impact of a parameter value on the final result was large, then the parameter was considered to be significant because the calculation was sensitive to changes in the parameter value. Based on the sensitivity analysis, the parameters were grouped into three categories: (1) sensitive parameters, (2) parameters with limited sensitivity, and (3) parameters not exhibiting sensitivity. We developed uncertainty distributions for the sensitive parameters identified using these categories. Of the more than 50 parameters evaluated, the sensitivity analysis, which will be described later in this report, identified the following parameters as critical:

- Mass loading factor
- Area of contamination
- Mean annual wind speed
- Distribution coefficients.

We emphasize these parameters in this report. Other parameters used in the calculation that were not sensitive in the analysis are identified but not discussed in detail. Parameter values that were not sensitive or marginally sensitive were not changed and are the same as those reported previously (DOE/EPA/CDPHE 1996). The only exceptions were thickness of the contaminated zone, depth of soil mixing layer, soil-to-plant transfer factors, irrigation water contamination fraction, external gamma shielding factor, and initial concentrations of radionuclides, where *RAC* determined that a different value was more appropriate based on the literature or site-specific data. *RAC* also selected the most current recommended dose conversion factors related to insoluble forms of plutonium.

Difference between Versions of RESRAD

The original calculations of soil action levels performed by DOE, EPA, and CDPHE used RESRAD Version 5.61 (DOE/EPA/CDPHE 1996). Since that time, the code developers have released updated versions of RESRAD. The most recent version of the code, Version 5.82, will be used for all independent calculations of soil action levels; therefore, we used it for the sensitivity analysis conducted for Task 3. Version 5.82 contains one major difference in an important pathway for the Rocky Flats calculations, and that difference focuses on the resuspension of soil. The calculation of air concentration of contaminated material has been adjusted to reflect the current understanding of resuspension. The change in the formulation of the area factor, sometimes called the enhancement factor, was discussed in detail in the Task 2 report. The impact of the change on the results of the DOE scenario calculations is discussed here.

Each scenario, dose level, and radionuclide was evaluated for the impact of this change in the code. With all parameter values held constant, the soil action levels predicted by RESRAD Version 5.82 were much higher than those predicted with older versions of the code. The single change in the formulation of the area factor in the RESRAD code predicted a significantly different dose via the resuspension pathway, reducing the relative importance of inhalation dose.

Because *RAC* believed inhalation to be of greater importance than indicated by the RESRAD calculations, we chose to develop our own formulation for resuspension. This is discussed in detail in a later section of this report, but the key characteristic of this new resuspension calculation is the use of site-specific data, namely soil and air concentration data.

SENSITIVITY ANALYSIS

To determine the parameters to be examined for uncertainty, we employed a single parameter sensitivity analysis. A single parameter analysis is defined by changing only one parameter at a time to analyze the impact of that change on the solution. This analysis was done earlier in the project for RESRAD Version 5.61 but was completed again using the current version of RESRAD, Version 5.82. Although an analysis of this sort ignored the possibility for correlation of parameters, we recognized this limitation and attempted to make concessions for it whenever possible.

A convenient feature of RESRAD Version 5.82 is a built-in sensitivity analysis tool. This tool allows the user to define a series of input values for a single parameter in the calculations. The user may multiply and divide the deterministic value of the parameter by any number to produce a stochastic range. The three values that define this range (minimum, median, and maximum) are used in the RESRAD calculations to calculate dose, dose to source ratio, and soil concentration for each pathway and each radionuclide, as well as the total dose from all sources. The code then produces graphics that reflect the range of calculation results using the range of input values.

For this sensitivity analysis, parameter values were allowed to vary by a factor of 10 in either direction (the median value was multiplied and divided by 10) unless the possible range of parameter values defined by RESRAD would be exceeded by this level of variation. In these cases, RESRAD defaults to the next largest factor that can be multiplied and divided into the median without exceeding the RESRAD limits.

While this method of evaluating sensitivity is certainly not without limitations, it did provide us with a good metric for evaluating change in the outcome of the calculation. The sensitivity analysis provided in RESRAD limits the user to evaluating some multiple (and divisor) of the defined median value. Varying the input value by a factor of 10 at least allowed us to evaluate the possible impact of the same degree of variation in any parameter on the outcome. We intended only to evaluate the model's sensitivity to change and did not intend to evaluate variability in the parameter. Variability in the parameter is defined in this task report and will be evaluated in the Task 5 report. *RAC* recognized the shortcomings of this sensitivity analysis but believed the analysis to be more than adequate for the purposes of this task report.

The results of this analysis fell into several categories. The parameters of primary importance have been identified as sensitive parameters. These parameters, when varied by a factor of 10, changed the output value of the calculation by more than a factor of 2. One exception to this was the area of contamination. The area of contamination, when varied by a factor of 10, changed the outcome of the calculation by less than a factor of 2. However, in our treatment of the resuspension calculation, the area of contamination was a parameter of increased importance. To treat resuspension on a site-specific basis, it was critical that area of contamination also be treated on a site-specific basis. In fact, our calculation, which will be explained in detail in a later section of this report, used contaminated area in a very important way. It is for this reason that we grouped this parameter with sensitive parameters in this report.

Another group of parameters showed limited sensitivity, but in several cases, the values were changed to reflect site-specific conditions. A parameter that showed limited sensitivity changed the outcome of the calculation by less than a factor of 2. Finally, a large fraction of the parameters did not exhibit any sensitivity to change. These parameters have been identified and the values, in general, were not changed from the value used in the DOE/EPA/CDPHE calculation.

Sensitive Parameters

The following parameters have a significant impact on the outcome of the calculation when values of the parameters are changed:

- Mean annual wind speed
- Area of the contaminated zone
- Distribution coefficients
- Mass loading.

These parameters were represented by either a distribution or a site-specific value based on other parameter distributions. These sensitive parameters are discussed in detail in a later section of this report titled “Uncertainty Distributions.”

Parameters with Limited Sensitivity

Another group of parameters showed some slight sensitivity to change. We selected either the previously used DOE/EPA/CDPHE value or a value more consistent with the literature. We justify the use of the values chosen below.

Cover Depth

The cover depth is the depth of soil that must be removed to reveal the contaminated zone. The value currently used in the calculation is 0 m, and any increase in the value for cover depth decreases estimated dose and increases soil action level. We believed that the use of this value was reasonable, and it was not changed.

Depth of Soil Mixing Layer

The depth of the soil mixing layer is the depth of surface soil available for resuspension. This depth represents that layer of soil within which contamination is uniformly distributed. This value is used to calculate the depth factor, which is the fraction of total resuspendible soil that is contaminated.

The research of Webb et al. (1997) showed that throughout the top 3 cm (0.03 m), contamination was primarily uniform, with perhaps a slight dip in contamination at lower depths. Webb et al. (1997) also provided a fractional contamination profile that allows total contamination in the top 3 cm (0.03 m) to be determined based on concentrations measured at other depths.

In the previous soil action level calculations (DOE/EPA/CDPHE 1996), the values for soil mixing layer and thickness of the contaminated zone were equal. *RAC* did not believe that setting the available depth for resuspension and the total thickness of the contaminated zone equal to each other was supported by the data from Rocky Flats. Based on the research of Webb et al.

(1997), *RAC* selected a value of 0.03 m for the depth of the soil mixing layer. We were, in fact, constrained to the use of this depth by the available soil concentration data.

Soil-to-Plant Transfer Factors

Soil-to-plant transfer factors quantify that portion of contamination in soil that is transferred to plants via root uptake. In January 1999, the National Council on Radiation Protection and Measurements (NCRP) issued Report No. 129, *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies* (NCRP 1999). This screening methodology suggests distributions for soil-to-plant transfer factor that reflect uncertainty resulting from different soil conditions, soil types, and soil chemistry. The values given in Report No. 129 (NCRP 1999) were adapted from values suggested in Report No. 123 (NCRP 1996) with application of uncertainty in the form of a geometric standard deviation. The values with their associated geometric standard deviations are shown in Table 1. These recommendations were not available at the time of the production of the DOE/EPA/CDPHE report. *RAC* believed that the use of these distributions enhanced the calculation, so they were selected to be used in the independent calculation for Task 5.

Table 1. NCRP Report No. 129 Soil-to-Plant Transfer Factor Values (in units of Bq kg^{-1} wet vegetation per Bg kg^{-1} dry soil)[a]

Element	Median soil-to-plant transfer factor	Geometric standard deviation
Plutonium	1.0 _ 10^{-3}	2.5
Americium	1.0 _ 10^{-3}	2.5
Uranium	2.0 _ 10^{-3}	2.5
Neptunium	2.0 _ 10^{-2}	2.5
Palladium	1.0 _ 10^{-2}	3.0
Lead	4.0 _ 10^{-3}	2.5
Radium	4.0 _ 10^{-2}	2.5
Actinium	1.0 _ 10^{-3}	3.0
Thorium	1.0 _ 10^{-3}	2.5

[a] Source: NCRP (1999).

Indoor Dust Filtration

The value of the indoor dust filtration factor represents the fraction of the total outside air contaminant concentration that is available indoors. A value of 1 means that the air contamination inside a building is equal to outdoor air contamination. *RAC* reviewed the available data on this parameter value, and there was a large degree of discrepancy among the available data. The values for this parameter vary widely among different studies. There are studies that suggest that this value could be as large as (or even larger than) 1, and other studies suggest it be no larger than 0.3. The NCRP has suggested that the best way to evaluate this parameter would be a site-specific study of indoor vs. outdoor air concentrations. Obviously, the time and resources of this project limit us from doing a study of this type. There is very little agreement within the literature for an appropriate value for this parameter. Because of this lack of agreement and the unknown future at the site, *RAC* did not change this value for our independent calculation, and we maintained the value of 1.0 used in the DOE/EPA/CDPHE calculation.

Irrigation Water Contamination Fraction

The value of the fraction of irrigation water contaminated by groundwater was 0.0 for the previous analysis (DOE/EPA/CDPHE 1996). As described in the scenarios section of this report, *RAC* has determined that there is a possibility that enough water exists and is accessible in the aquifer to provide at least limited drinking and irrigation water. To perform an accurate analysis, that irrigation water must be considered contaminated. The value for the contamination fraction of the irrigation water for this analysis was set to 1.0, implying that the irrigation water is as contaminated as the groundwater. If we assumed that irrigation water came directly from the aquifer, this implication was reasonable, and a value of 1.0 was justified.

Thickness of Contaminated Zone

The thickness of the contaminated zone represents the vertical distance over which radionuclide contamination levels are clearly above background. This differs from the depth of soil mixing layer in that over the contaminated zone, it is not required that the contamination be uniform. Changes in this parameter do influence the outcome of the calculation somewhat, but this value has been well characterized at Rocky Flats. The research of Webb et al. (1997) indicated that contamination was distributed over the top 20 cm (0.2 m) of soil, with very little movement of that soil within the column over the past 20 years. For this reason, we treated the parameter deterministically and used a value of 0.2 m.

Parameters not Exhibiting Sensitivity

A large fraction of the parameters required for using RESRAD showed no sensitivity to change in their values. Although no sensitivity was shown, in some cases *RAC* has determined that a different values is more appropriate for use in the RESRAD calculations based on site-specific data or literature values.

External Gamma Shielding Factor

For external gamma shielding factor, *RAC* decided to use a more traditional definition of the parameter to select a value. The external gamma shielding factor (*EGS*) is the ratio of the external gamma radiation level indoors to the level outdoors. This value is used in the RESRAD code to calculate occupancy factor as shown in Equation (1).

$$\text{Occupancy factor} = \frac{\left(\text{h d}^{-1}\text{ indoors}\right)}{24\text{ hours}} \times EGS + \frac{\left(\text{h d}^{-1}\text{ outdoors}\right)}{24\text{ hours}} \times 1.0 + \frac{\left(\text{h d}^{-1}\text{ offsite}\right)}{24\text{ hours}} \times 0.0 \qquad (1)$$

The occupancy factor is then used in calculations of dose from the external gamma pathway by determining the total external gamma exposure during the course of a day.

The RESRAD default value for this parameter is 0.7. The values used in the previous calculations for the resident, open space user, and office worker were 0.8, 0.014, and 0.17, respectively (DOE/EPA/CDPHE 1996). The fraction of time spent indoors for all three scenarios was defined as 1.0, so these values were developed to represent the occupancy factor.

This use of the external gamma shielding factor to represent occupancy was unnecessary because RESRAD performs that calculation when given the appropriate parameter values. *RAC* has chosen to use the gamma shielding factor for its intended purpose and to define fractional time indoors/outdoors/offsite as a part of the exposure scenarios. This allows RESRAD to calculate occupancy as it is designed to do, making the parameter valuation easier to use and understand.

The external gamma shielding factor selected by *RAC* was 0.7. This will be used by RESRAD in combination with the time spent indoors, outdoors, and offsite to calculate occupancy factor as shown below for the *RAC* residential rancher.

$$\text{Occupancy factor} = \left(\frac{10 \text{ h outdoors}}{24 \text{ h d}^{-1}}\right) \times 1.0 + \left(\frac{14 \text{ h indoors}}{24 \text{ h d}^{-1}}\right) \times 0.7 = 0.825 \qquad (2)$$

This methodology was more straightforward and consistent with the intended parameter use in RESRAD. *RAC* recommended the value of 0.7 for this parameter and has defined fraction of time indoors, outdoors, and offsite as a part of the scenarios described later in this report.

Initial Concentration of Radionuclides

Initial concentrations of radionuclides are important values to define when discussing dose as an endpoint. The existing DOE/EPA/CDPHE calculation defined initial concentrations of each radionuclide of interest (^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, ^{241}Am, ^{234}U, ^{235}U, and ^{238}U) as 100 pCi g^{-1} (3700 Bq kg^{-1}). Although the soil action levels produced by RESRAD are not dependent on initial concentration, the results of the RESRAD dose calculation are meaningful only when values that represent actual concentrations in soil are used.

RAC used the available literature in combination with measured soil concentration data to produce actual concentrations in soil, initialized at the year that the soil action level calculations begin. A number of studies have characterized the ratios of contaminants in the Rocky Flats environment to one another. The literature listed relative mass percentiles of plutonium isotopes in 1971 (Krey et al. 1976) and relative concentration ratios of uranium isotopes and americium to ^{239}Pu in approximately 1993 (Litaor 1995). We converted these mass values to activities and allowed them to decay (or grow in, in the case of ^{241}Am) to the year 1999 for use in the RESRAD calculations. The relative concentrations of radionuclides derived from these studies are shown in Table 2. The values shown are relative to $^{239+240}$Pu (given a value of 1), and will be used to calculate estimates of concentrations of each radionuclide for the current concentrations of $^{239+240}$Pu.

Table 2. Relative Concentrations of Radionuclides in Soil at Rocky Flats in 1999

Radionuclide	Relative concentration (to $^{239+240}$Pu)
^{238}Pu	0.0132
^{239}Pu	0.843
^{240}Pu	0.157
^{241}Pu	0.798
^{242}Pu	7.62 _ 10^{-6}
^{241}Am	0.111
^{237}Np	7.86 _ 10^{-7}

The current value for ^{239}Pu contamination varies spatially. *RAC* has identified contours of contamination levels using soil concentration data from Litaor et al. (1995), Litaor and Zika (1996), Webb et al. (1997), Illseley and Hume (1979), Ripple et al. (1994), Krey et al. (1976), and the CDPHE. We develop and present these contours in a later section of this report.

Uranium concentrations are more difficult to determine. Available data suggested that uranium exists on the Rocky Flats site in a few small "hot spots." Determining where those hot spots might exist within the scope of this study is difficult.

Litaor (1995) looked at the extent and distribution of uranium in the Rocky Flats environment. Litaor discovered that the uranium followed no recognizable spatial distribution pattern and was not in concentrations readily discernible from background, for the most part. The RFP contribution to ^{234}U was determined to be negligible. The elevated soil concentrations of ^{235}U were localized to an area east of the industrial section. Litaor suggested that these concentrations might have resulted from surface flow and interflow from the east spray field (Litaor 1995). The ^{238}U activities that were the highest were located in the immediate vicinity of the 903 Area, but they did not extend beyond that area, suggesting that uranium was not dispersed in the same way as plutonium. Litaor suggested that this is likely due to the differences in the solubility characteristics of the two nuclides.

Even with the few elevated concentrations of uranium, the concentrations of ^{234}U and ^{235}U were generally well within the natural range for uranium isotopes in soil. Only ^{238}U showed elevated concentrations in Litaor's study area, and those were located immediately around the 903 Area.

It is likely that the most significant uranium concentrations would exist in locations where uranium was stored or burned, such as the trenches, or perhaps in solar pond sediments. Uranium contamination is definitely site-specific and would be above background only at a limited number of locations as dictated by Rocky Flats operations and disposal practices. Certainly, uranium is not distributed in any recognizable spatial pattern, and uranium contamination probably only exists in hot spots. The extent, concentration, and location of these hot spots are important for calculating any contribution to dose from uranium.

For our calculations of soil action levels for uranium, we selected a single location for which concentrations might be at a maximum and determined an action level for that location. This guides us to a better understanding of uranium and its potential risk to those at the Rocky Flats location. These calculations will be accomplished and outlined in the Task 5 report.

Plutonium Solubility and Dose Conversion Factors

Results from ongoing Actinide Migration Studies (AMS) at the site are helping to characterize the chemical and physical form of plutonium at the Rocky Flats site. The plutonium that is found in Rocky Flats soil is generally highly insoluble and attached to soil particles. This view is supported by the AMS, which show the effectiveness of the retention ponds in removing suspended solids and associated plutonium (and americium) from site surface water (RMRS 1998). Much of the plutonium discharged to Pond C-2 settles out of the water column, and plutonium concentrations measured further downstream in Woman Creek are an order of magnitude lower. In contrast, the ponds are less effective at removing uranium from the water column. This is expected because uranium has a higher solubility than plutonium and is more susceptible to dissolution and transport in the solution phase.

Recent work by researchers at the Los Alamos National Laboratory has characterized plutonium in samples from the 903 Area. Using powerful, new state-of-the-art analytical techniques, they have demonstrated that plutonium from under the asphalt pad at the 903 Area is insoluble PuO_2. The plutonium/americium ratio also indicates insoluble plutonium. These new results tend to confirm that plutonium in the soil at Rocky Flats is insoluble PuO_2 and, thus, may not get into the groundwater. While results from some of the AMS indicate that this insoluble form of plutonium may not enter groundwater, we are including the groundwater pathway in the rancher scenario. We do recognize, however, that our assessment of the groundwater pathway is limited by the pathway's complexity.

Plutonium mobility is another area under investigation by the AMS researchers that may play an important role at the site. One situation that may result in increased plutonium mobility is during extraordinary precipitation events in which the soil is saturated for significant amounts of time (Litaor and Zika 1996). Such conditions may result in subsurface storm flow, which is rapid, saturated, near-surface lateral flow from hill slopes that can discharge to seeps and streams because the groundwater is moving rapidly at a shallow depth. Subsurface storm flow is a potentially important pathway for plutonium in localized surface soil contamination areas where shallow or perched groundwater discharges to seeps or stream channels.

These solubility studies allow dose conversion factors to be determined for plutonium and other radionuclides. Insoluble forms of plutonium would be classified as slow clearance materials. In ICRP 30 (ICRP 1978), these forms of plutonium were classified as clearance type Y. *RAC* has researched the most updated values available for dose conversion factors from ICRP (1999). Clearance classification has changed somewhat. Instead of identifying clearance based on time it takes to clear the material (D, W, or Y to represent days, weeks, or years), ICRP identified the clearance by rate at which material is cleared (F, M, or S to represent fast, medium, or slow). These classifications are generally interchangeable on a respective basis, so insoluble plutonium would now be classified as type S. Table 3 shows the most recent values for inhalation and ingestion dose conversion factors in comparison to the values from ICRP 30 for the radionuclides of interest at Rocky Flats.

Table 3. Dose Conversion Factors (DCFs) for Independent Calculation (mrem pCi^{-1})[a]

Radio-nuclide	ICRP 30[b] clearance class	ICRP 30 Inhalation DCF	ICRP 71[c] clearance class	ICRP 71 Inhalation DCF	ICRP 30 f_1	ICRP 30 Ingestion DCF	ICRP 67[d] f_1	ICRP 67 Ingestion DCF
^{241}Am	W	0.444	M	0.155	0.001	0.00364	0.0005	0.00074
^{238}Pu	Y	0.288	S	0.059	0.00001	0.0000496	0.0005	0.00085
^{239}Pu	Y	0.308	S	0.059	0.00001	0.0000518	0.0005	0.00093
^{240}Pu	Y	0.308	S	0.059	0.00001	0.0000518	0.0005	0.00093
^{241}Pu	Y	0.00496	S	0.00063	0.00001	0.00000077	0.0005	0.00002
^{234}U	Y	0.132	S	0.035	0.05	0.000283	0.02	0.00018
^{235}U	Y	0.123	S	0.031	0.05	0.000267	0.02	0.00017
^{238}U	Y	0.118	S	0.030	0.05	0.000269	0.02	0.00017

[a] The units of mrem pCi^{-1} are the conventional units used in RESRAD. To convert to standard units of Sv Bq^{-1}, simply divide the value in the table by 3700.
[b] ICRP 30 values have been used in RESRAD Versions 5.61 and 5.82.
[c] ICRP 71 listed the latest inhalation dose conversion factors (also given on ICRP CD-ROM [ICRP 1999]).
[d] ICRP 67 listed the latest ingestion dose conversion factors (also given on ICRP CD-ROM [ICRP 1999]).

Dose conversion factors do exhibit some limited age dependency. For very young babies (0–3 months), f_1[a] values for ingestion are as much as 10 times higher than the adult values, increasing the dose conversion factor by about 16 times. All other ages have ingestion dose coefficients somewhat less than a factor of 2 higher than the adult values.

The dose conversion factor values have changed rather significantly since the last ICRP publication. There are a number of reasons for these changes.

For inhalation dose conversion factors, changes in the respiratory tract model have the largest effect on the differences. The new respiratory tract model indicates reduced uptake from the lung. For an aerosol with an activity median aerodynamic diameter of 1 μm, the new model indicates roughly 50% of the inhaled activity deposited in the tract, in contrast with the 63% predicted by the old model. This distinction results from the new model being characterized as a nose breather, where a large fraction of the inhaled activity would be deposited in the anterior regions of the nasal passage and would never make it to the gastrointestinal tract to be adsorbed. The difference in deposition between the two models is almost a factor of two.

There is also a new model for the fraction of the lymph node irradiation attributed to lung dose, as well as a new model for the behavior of plutonium once it enters the blood stream, considering the movement of plutonium from bone surfaces into bone volume. All of these factors contribute to lowering the absorbed dose from inhalation of unit activity of plutonium.

The ingestion dose conversion factors reflect the difference introduced by the changes in the behavior of plutonium in the blood stream, as well as differences in new tissue weighting factors and adsorption coefficients (Eckerman 1999).

[a] f_1 is a factor that defines the retention of radionuclides in the body. The higher the value of f_1, the greater the retention.

Remaining Parameters

The outcome of the calculation was not sensitive to changes in the following parameter values:

- Nearly all of the saturated zone parameters (excluding K_d)
- All of the uncontaminated zone parameters
- Nearly all of the contaminated zone parameters including evapotranspiration coefficient, erosion rate, porosity, conductivity, density, b parameter, precipitation, irrigation rate and mode, and runoff coefficient
- Length parallel to the aquifer
- Watershed area
- Storage times for food
- Mass loading for foliar deposition
- Plant contamination fraction
- Thickness of the unsaturated uncontaminated zone
- Water table drop rate
- Well pump intake depth
- Well pumping rate.

Because of the insensitivity of the calculation to changes in these parameter values, we determined that additional work characterizing these values was not justified. In all cases, we accept and will use the values suggested in the original soil action level document (DOE/EPA/CDPHE 1996). In two cases, DOE used different values for the same parameter in each of the three scenarios in the existing soil action level calculations (DOE/EPA/CDPHE 1996). These parameters were irrigation rate and evapotranspiration coefficient. Neither of these parameters were found to be very sensitive to change. *RAC* used the values selected in the DOE/EPA/CDPHE calculations for the hypothetical resident scenario (DOE/EPA/CDPHE 1996).

Some of these remaining parameters that were not sensitive to change are part of the drinking/groundwater calculation, and they have no impact on the current soil action level calculation (DOE/EPA/CDPHE 1996) because none of the scenarios include the drinking water pathway. We explore the impact of this pathway in the following section of this report. Table 4 compares the parameter values to be used in the independent calculation to the DOE/EPA/CDPHE values. For more information on the distributions and values for the sensitive parameters, refer to the section titled "Uncertainty Distributions."

Table 4. Parameter Values to be Used in the Independent Calculation

Parameter name	DOE value	*RAC* value
Sensitive Parameters		
Distribution coefficient	Pu = 218 $cm^3\ g^{-1}$ (or $L\ kg^{-1}$) Am = 76 $cm^3\ g^{-1}$ U = 50 $cm^3\ g^{-1}$	Treated stochastically based on Rocky Flats measurements and other available data
Area of contaminated zone	40,000 m^2	Defined based on soil concentration measurements
Mass loading	0.000026 $g\ m^{-3}$	Model calibrated based on results of soil and airborne concentration analysis
Mean annual wind speed	Not required for RESRAD V 5.61	Used annual average wind data collected over 5 years
Limited Sensitivity Parameters		
Thickness of contaminated zone	0.15 m	0.20 m
Inhalation shielding factor	1.0	1.0
Soil-to-plant transfer factors	Deterministic Pu = 1.0 _ 10^{-3} Am = 1.0 _ 10^{-3} U = 2.0 _ 10^{-3}	Treated stochastically based on NCRP 129 recommendations
Cover depth	0 m	0 m
Irrigation water, contamination fraction	0	1.0
Depth of soil mixing layer	0.15 m	0.03 m
Parameters Not Exhibiting Sensitivity		
Initial concentrations of radionuclides	100 $pCi\ g^{-1}$	Based on soil concentration measurements by Webb et al. (1997), Litaor (1995), Illsley and Hume (1979), CDPHE (as deposited by Litaor), and Krey et al. (1976)
External gamma shielding factor	0.8 – residential 0.014 – open space 0.17 – office worker	0.7 – for all scenarios, indoor/outdoors time fractions will describe occupancy
Density of contaminated zone	1.8 $g\ cm^{-3}$	1.8 $g\ cm^{-3}$
Contaminated zone erosion rate	0.0000749 $m\ y^{-1}$	0.0000749 $m\ y^{-1}$
Contaminated zone total porosity	0.3	0.3
Contaminated zone effective porosity	0.1	0.1
Contaminated zone hydraulic conductivity	44.5 $m\ y^{-1}$	44.5 $m\ y^{-1}$
Contaminated zone b parameter	10.4	10.4
Evapotranspiration coefficient	0.253 – residential 0.920 – open space, office worker	0.253

Table 4. (Continued)		
Parameter name	DOE value	*RAC* value
Precipitation rate	0.381 m y^{-1}	0.381 m y^{-1}
Irrigation rate	1.0 m y^{-1} – residential 0 m y^{-1} – open space, office worker	1.0 m y^{-1}
Irrigation mode	Overhead	Overhead
Runoff coefficient	0.004	0.004
Watershed area	8,280,000 m^2	8,280,000 m^2
Accuracy for water/soil computations	0.001	0.001
Density of uncontaminated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Uncontaminated zone total porosity	0.3	0.3
Uncontaminated zone effective porosity	0.1	0.1
Uncontaminated zone hydraulic conductivity	44.5	44.5
Uncontaminated zone b parameter	10.4	10.4
Density of saturated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Saturated zone total porosity	0.3	0.3
Saturated zone effective porosity	0.1	0.1
Saturated zone hydraulic conductivity	44.5	44.5
Saturated zone hydraulic gradient	0.15	0.15
Water table drop rate	0 m y^{-1}	0 m y^{-1}
Well pump intake depth	10 m	10 m
Nondispersion/mass balance	Nondispersion	Nondispersion
Well pumping rate	250 m^3 y^{-1}	250 m^3 y^{-1}
Thickness of uncontaminated, unsaturated zone	3 m	3 m
Length parallel to aquifer flow	200 m	200 m
Elapsed time of waste placement	0 y	0 y
Dilution length	3 m	Not required for RESRAD Version 5.82
Shape factor	Circular	Based on results of soil concentration analysis
Plant food, contamination fraction	1.0	1.0
Drinking water, contamination fraction	Not used	1.0
Mass loading for foliar deposition	0.0001 g m^{-3}	0.0001 g m^{-3}
Depth of roots	0.9 m	0.9 m
Groundwater fractional usage, irrigation	1.0	1.0
Average storage time for fruits, nonleafy vegetables, and grain consumption	14 d	14 d
Average storage time for leafy vegetable consumption	1 d	1 d
Average storage time for well water and surface water use	1 d	1 d

The Groundwater/Drinking Water Pathway

Groundwater is an extremely complex pathway (described in Task 2), and *RAC* will not assess it in significant detail in the soil action level project because of the extensive ongoing research and the complexity of the interacting processes. We will, however, provide bounding level, screening calculations for the rancher-based scenarios with contaminated drinking water as a pathway for dose. The intent of doing this is not to provide quantitative results but rather to assess the potential importance of the drinking water pathway and provide a mechanism for making calculations when groundwater parameters have been more accurately determined.

For the drinking water pathway, as it will be used in these calculations, the contaminated fraction of drinking water is 1.0; that is, 100% of the receptors' drinking water comes from contaminated groundwater and is as contaminated as the groundwater. By setting the drinking water contamination equal to that in the groundwater, we protect receptors from groundwater resources near their source, thus, protecting the resource at farther downgradient locations.

To explore the sensitivity of the drinking water pathway, we used a deterministic calculation of dose. The parameter values for the five sensitive parameters identified above were not changed from those used in the previous analysis (DOE/EPA/CDPHE 1996) for this sample calculation. For the remaining parameters, we used the values defined in Table 4, the scenario parameters associated with the previous analysis' hypothetical resident, the initial concentration ratios defined in Table 2, and an initial concentration of ^{239}Pu of 500 pCi g^{-1} (18,500 Bq kg^{-1}). This definition of initial concentrations is important in this analysis because we will use dose as the endpoint for comparison.

The maximum annual dose from all radionuclides calculated without the inclusion of the drinking water pathway was 29 mrem y^{-1} (0.29 Sv y^{-1}) at time t = 0. The maximum dose, including the drinking water pathway, was 117 mrem y^{-1} (1.17 Sv y^{-1}) at time t = 221 years. This dose is primarily from drinking water ingestion.

The increase in dose when the drinking water pathway is included is significant. It is important to understand several things about this calculation. First, the increase in dose was due almost entirely to dose from ^{241}Am as it reached the groundwater. The amount of time it took for the americium to reach the groundwater was dependent on the numerical value of the soil-water equilibrium distribution coefficient, which describes the partitioning of contaminants between solid and aqueous phase. This parameter value is critical when the groundwater model is a simple linear model, as it is in RESRAD. If the value of the distribution coefficient is greater than about 220 cm^3 g^{-1}, the nuclide will not reach the groundwater during the 1000-year RESRAD simulation. Based on the RESRAD conceptual model for subsurface transport and the hydrologic transport parameter used in the simulation, it takes over 200 years for significant concentrations of the americium to reach the groundwater and be available in the drinking water, using the DOE distribution coefficient value of 76 cm^3 g^{-1} for ^{241}Am. This calculation was completed only for illustrative purposes, to demonstrate the potential importance of the groundwater pathway. Distribution coefficient is revisited later in this report.

However, much is unknown about the mechanisms by which americium and other radionuclides are transported through the soil column and into the aquifer. There is an additional degree of uncertainty about the properties of the aquifer. Studies on the mobility of radionuclides in the Rocky Flats environment do reveal some important information. Both plutonium and americium are strongly adsorbed, limiting their mobility considerably. The distribution coefficients indicated by research are quite high for both americium and plutonium at Rocky Flats, indicating a high affinity for the solid phase. Parameters that describe the distribution coefficient, bulk hydrologic properties of the subsurface, and precipitation and infiltration in RESRAD dictate the rate at which radionuclides are transported into the aquifer and, therefore, control the calculation of dose from the drinking water pathway.

The vertical distribution of radionuclides in soil is another indicator of mobility, and this has been described by a number of researchers. Some convincing evidence comes from Webb (1996), which revisited the Rocky Flats study documented in Little (1976) and found that the vertical distribution of plutonium and americium has remained nearly the same over the last 20 years. This vertical distribution decreases with depth in the soil column.

There is, however, a recognized potential for transport of radionuclides attached to small colloid-sized particles. Attachment and subsequent transport of these particles would significantly enhance mobility because they do not behave as a dissolved phase species in terms of their sorption-desorption properties. DOE qualitatively looked at the possibility of this transport in their Resource Conservation and Recovery Act facility investigation/remedial investigation Operable Unit-2 (OU-2) document (DOE 1995a). In the DOE report, a study by Penrose et al. (1990) was cited. The Penrose study suggested that small colloids (<0.45 μm) could transport plutonium and americium over large distances in the subsurface. However, colloids larger than 0.45 μm are basically immobile under the same conditions that made small colloid transport possible. Analytical groundwater data from OU-2 for filtered (with a 0.45-μm filter) and unfiltered samples were compared. These data suggested that most of the plutonium and americium in groundwater was associated with the unfiltered sample and, therefore, with particles larger than 0.45 μm in diameter. This qualitative analysis seems to indicate that colloidal transport is not a mechanism by which significant quantities of plutonium and americium are transported to the groundwater at Rocky Flats.

Other studies suggest the opposite is true. Kersting et al. (1999) looked at the possibility for colloidal transport in groundwater at the Nevada Test Site. The researchers observed that radionuclide concentrations in groundwater were associated with the colloidal fraction, and they showed the plutonium source to be an underground nuclear test site 1.3 km away from the groundwater well.

Honeyman (1999) agreed that colloidal transport was certainly a potential and probable mechanism for radionuclide transport, but it pointed out the flaws in the Kersting study. Honeyman recited the three conditions that must be met for colloidal transport to be defensibly proved: (1) colloids must be present in the groundwater, (2) contaminants must associate with the colloids, and (3) the combination of the colloid and contaminant must move through the aquifer. Kersting et al. proved only the first two of these three conditions to be true in their study. In fact, Kersting et al. pointed out the possibility that the study conditions (i.e., increased well pumping) may have enhanced colloidal concentration, preventing quantification of the colloidal load.

The importance of the above discussion is to point out that, at the present time, very little is understood about the mechanisms of colloidal transport of radionuclides in groundwater aquifers. Evidence seems to show that this transport mechanism may be important, but this is an area of current research. Applying any detailed model requires field investigations of the site hydrology and a modeling effort that spans several years to calibrate model results with field measurements.

We looked at the significance of the groundwater/drinking water pathway in this document in terms only of its *potential* for dose. Any dose values resulting from drinking water pathway calculations cannot be finalized during the course of this project simply because the pathway is far more complex than its representation in RESRAD and neither the transport properties nor the aquifer properties are understood at Rocky Flats.

What we learned from this analysis is that groundwater can have an impact on dose that needs to be recognized. Because of the severe limitations on time and resources in this study, we can only recommend that a future study be directed toward this type of work, particularly looking at the migration of ^{241}Am and its progeny.

UNCERTAINTY DISTRIBUTIONS

In this project, the term uncertainty usually implies lack of knowledge about the value of a model parameter or the accuracy of a model prediction. We represent these uncertainties as probability distributions. This lack of knowledge about a parameter value can arise from (a) variability of the parameter over space or time, (b) variability among different experiments or field studies that measure the parameter, or (c) variability within individual studies in which measurements, by design, are taken under different sets of controlled conditions. If the data available to us correspond to times, locations, or conditions other than those relevant to this study, then the variability within our limited data (expressed, for example, by the sample standard deviation) may not adequately reflect the uncertainty of the estimates.

Some environmental parameters are difficult to observe directly, and estimates must be based on inferences from available observations of other presumably correlated quantities. But such an indirect approach usually relies on a model connecting the desired quantity with the ones being measured, and use of the idealized model usually introduces uncertainties of its own. An example relevant to Rocky Flats is resuspension. Factors for wind-driven resuspension have been calculated as the ratio of the air concentration of a contaminant (e.g., becquerel of plutonium per cubic meter) divided by the amount of contaminant per square meter of soil (the soil measurement is taken to a depth that is considered resuspendable). A resuspension factor (per meter) is multiplied by a measured soil concentration of a contaminant (e.g., becquerels per square meter) to predict an airborne concentration of the contaminant (becquerels per cubic meter). The implied model assumes a large source area of soil that is uniformly contaminated and uniform in those properties that affect the mechanisms of resuspension (e.g., ground cover, soil particle size distributions, moisture, depth of the resuspendable layer, and terrain topography). It is also assumed that the resuspension factor represents airborne concentrations that are averaged over a sufficient period to be characteristic of the local meteorological conditions. Such uniformities are seldom available to field studies (or applications), and measurements of factors for wind-driven resuspension range from 10^{-4} to 10^{-11} m^{-1} (Sehmel 1972). Without other information, this range is an indication of uncertainty for the local resuspension factor. The resuspension factor for a contaminated location also changes over time as the contaminant migrates downward into soil or undergoes superficial erosion. Anspaugh et al. (1975) and others have made generic characterizations of this temporal trend for plutonium resuspension factors.

Even if direct measurements of the desired quantity are available, they may have been made at a time other than the one relevant to the application. For example, meteorological predictions for environmental assessments often use a joint frequency table of wind speed, wind direction, and atmospheric stability based on five consecutive years of hourly observations at a given location. But when the time of interest for predictions is not within the 5-year period, use of this frequency table introduces a component of uncertainty that results from the variability of the meteorological frequencies over time. This component can be as much as a factor of 2 in predicted annual-average air concentrations, and it is not the only component of uncertainty in such predictions.

In this report, we propose distributions of uncertainty for various parameters that are inputs to RESRAD. To make predictions that reflect these uncertainties, we sampled values for the affected set of RESRAD parameters from these probability distributions, ran RESRAD to calculate the outcome, stored the outcome, and repeated the cycle many times, sampling from the assumed distributions each time. The set of results forms a distribution of outcomes that represents the propagated parameter uncertainties. This distribution might represent dose, dose-to-source ratio, or soil concentration/action level.

The parameters emerging from the sensitivity analysis as important for these calculations were area of contaminated zone, distribution coefficient, mass loading, and mean annual wind speed. As the most critical parameters, it was important to develop distributions of values, where appropriate, using a combination of site-specific data and information from the open literature. This section describes the treatment of these parameters for the independent calculation.

Distribution Coefficient

The transport of radionuclides in groundwater involves solving two fundamental equations that describe a) movement of water within the geologic media, and b) movement of the dissolved constituents (radionuclides). Movement of water is typically described by quantifying water fluxes and velocities (which are functions of the hydrologic properties of the system and the level of saturation) in the system and must be determined first before proceeding with the contaminant transport calculations. Movement of water in porous media, particularly in unsaturated and fractured media, is an area of ongoing research, and much of the overall uncertainty related to groundwater models can be attributed to lack of understanding and poor characterization of these processes. Assuming these processes have been adequately characterized, we then apply the contaminant transport equations to calculate concentrations of radionuclides in pore water at a selected receptor location. Most radionuclides form two phases in groundwater; a dissolved phase that travels with the water, and a sorbed phase that remains attached to the porous matrix. The degree at which a radionuclide sorbs depends on the chemistry of the pore water, the porous media, and the radionuclide itself. At relatively dilute concentrations, the ratio of the concentration in the attached or sorbed phase to that in the pore water remains constant at equilibrium. This ratio defines the linear sorption or distribution coefficient (K_d) and is given by

$$K_d = \frac{C_s}{C_w} \tag{3}$$

where
C_s = the concentration of radionuclide sorbed onto the porous matrix (Ci g^{-1})
C_w = the concentration of the radionuclide in the pore water (Ci mL^{-1})

The distribution coefficient relationship is assumed to be valid over the ranges of concentrations encountered in the environment. In addition, sorption reactions are assumed to occur quickly and achieve equilibrium conditions over the time spans considered (1 to 1000 years). In reality, the sorption process is much more complicated than suggested by the simple distribution coefficient, and is an area of ongoing research. Much of the uncertainty associated with groundwater transport calculations may be attributed to the simplistic treatment of sorption processes. However, without substantially greater resources and time, there is little we can do but resign ourselves to using the distribution coefficient approach in our simulations.

Sorption reactions have the net effect of slowing down or retarding the movement of radionuclides in groundwater. The higher the distribution coefficient, the higher the degree of sorption and the slower the contaminant moves in groundwater. If the radionuclide is non-reactive, that is, it does not sorb and remains entirely in the aqueous phase, its average velocity in groundwater is the same as the water.

Values for K_d vary greatly with physical and chemical properties of the solid, liquid, and radionuclide. Distribution coefficients tend to be greater for finer-grained materials such as silt and clay compared to coarser materials like sand or fractured igneous rocks because the finer materials have cation-exchange capacity. Generally constant for a system under specified conditions, the value for K_d can range over orders of magnitude for different situations, and many of these different situations may exist in the strata of different geologic properties that underlie a given aboveground area. Consequently, the K_d tends to be one of the more sensitive parameters in any calculation involving groundwater.

Values for K_d have been predicted for plutonium, uranium, and americium in the environment around the 903 and Mound Areas (DOE 1995a). The Actinide Migration Studies Panel initiated measurements of K_d in a limited portion of the Rocky Flats environment for uranium and plutonium. Distribution coefficients have also been reported in the literature for a variety of environments. We used all this information to derive a probability distribution for K_d values in the Rocky Flats environment.

The values for the K_d used in the DOE/EPA/CDPHE soil action level document were derived from data reported by Dames and Moore (1984). Dames and Moore reported a range of values for the retardation factor from the literature. The retardation factor is derived from the contaminant mass balance in porous media.

$$C_T = C_w \theta_w + K_d C_w \left(1 - \left(\theta_w + \theta_a\right)\right)\rho_s \quad (4)$$

where
C_T = radionuclide concentration in the porous media (Ci mL^{-1})
C_w = radionuclide concentration in the water phase (Ci mL^{-1})
θ_w = water filled porosity
θ_a = air filled porosity
(s = particle density (g mL_1)
Kd = distribution coefficient (mL g_1)

Assuming the total porosity is equivalent to the effective porosity, and relating the bulk density ((b) to the particle density [(b = (s(1-((a +(w))], Equation 4 can be solved for Cw giving

EMBED Equation.3 (5)

The term, 1+Kd (b/(w represents the retardation factor (R). Solving to Kd yields

EMBED Equation.3 (6)

where
K_d = distribution coefficient (cm^3 g^{-1})

R = retardation factor
θ_w = effective porosity of the aquifer
ρ_b = bulk soil density (g cm^{-3}).

For the DOE determination of K_d, the values for θ_w and ρ_b were 0.10 and 1.84 g cm^{-3}, respectively. These values were measured for OU-2 and represent a reasonable estimate of site-specific parameters (DOE 1995a). The Dames and Moore (1984) retardation factor values for sand and clay soils are shown in Table 5 for each radionuclide, along with the associated K_d value calculated using equation (6).

Table 5. Dames and Moore (1984) Reported Retardation Factor Values and Calculated Distribution Coefficients

	Sand		Clay	
Radionuclide	R	K_d (cm^3 g^{-1})[a]	R	K_d (cm^3 g^{-1})
Americium	300	16.3	2500	136
Plutonium	840	45.6	7200	391
Uranium	840	45.6	7200	391

[a] The use of the K_d units of cm^3 g^{-1} are RESRAD driven. These units are equivalent to L kg^{-1}

DOE/EPA/CDPHE (1996) used the midpoint of the ranges shown in Table 5 for americium and plutonium to represent the K_d values for their calculations. Sheppard and Thibault (1990) reviewed a number of distribution coefficient measurements and produced ranges of K_d values for sand, loam, and clay soils. These ranges are shown in Table 6.

Table 6. Ranges of Distribution Coefficients from Sheppard and Thibault (1990) (in units of cm^3 g^{-1} or L kg^{-1})

	Sand		Loam		Clay	
Radionuclide	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
Americium	8.2	300,000	400	48,309	25	400,000
Plutonium	27	36,000	100	5,933	316	190,000
Uranium	0.03	2,200	0.2	4,500	46	395,100

Till and Meyer (1983) reported values for K_d for a variety of nuclides, but of the radionuclides of interest to this study, they showed values only for uranium. Table 7 shows the range of K_d values reported in their work.

Table 7. Range of K_d for Uranium Reported in Till and Meyer (1983)

Type of soil, uranium oxidation, and pH	K_d ($cm^3\ g^{-1}$ or $L\ kg^{-1}$)
Silt loam, U(VI), Ca-saturated, pH 6.5	62,000
Clay soil, U(VI), 5mM $Ca(NO_3)_2$, pH 6.5	4400
Clay soil, 1 ppm UO^{+2}, pH 5.5	300
Clay soil, 1 ppm UO^{+2}, pH 10	2000
Clay soil, 1 ppm UO^{+2}, pH 12	270
Dolomite, 100-325 mesh, brine, pH 6.9	4.5
Limestone, 100-170 mesh, brine, pH 6.9	2.9

The Actinide Migration Studies were established to specifically study different aspects of actinide migration and transport in the Rocky Flats environment. In a paper submitted to the panel, Honeyman and Santschi (1997), values of K_d for uranium and plutonium were reported. The authors cautioned that the data presented in their paper represented an upper range of likely values and that another study to determine the lower range of likely values needed to be completed.

Plutonium K_d values were measured in 903 Area lip soils. Uranium values were measured only for the oxidation state U(VI). Uranium geochemistry reveals that the U(VI) oxidation state is the most stable of the three most common oxidation states (U[IV], U[V], and U[VI]) and would also be the most mobile of these three states. Uranium K_d values were measured in solar pond core sediments. Table 8 presents the range of values measured.

Table 8. Range of Maximum K_d Values ($cm^3\ g^{-1}$ or $L\ kg^{-1}$) Measured at Rocky Flats by Honeyman and Santschi (1997)

Radionuclide	Range of possible maximum values
U(VI)	31.2–171
$^{239,240}Pu$	0.98 _ 10^4–1.16 _ 10^5

More than a factor of 5 difference exists between the values measured for uranium, and an order of magnitude exists between the values measured for plutonium. Again, these ranges reflect likely maximum values for K_d. The other data presented here show even larger ranges of values of K_d, which probably more accurately reflect the range of total possible values.

From the data presented in Tables 5-8, it is obvious that K_d values are highly variable and tend to be higher for finer-grained material (clay and silt) compared to coarser grained sands. Also, plutonium and americium K_d values tend to be higher than those for uranium.

RAC has created a distribution of K_d values for uranium, plutonium, and americium. These distributions of K_d values reflect the wide range of variability possible in K_d, giving careful consideration to the Honeyman and Santschi (1997) data set, indicating potential maximum values for K_d in the Rocky Flats system. Although data from Till and Meyer (1983) presented in Table 7 show much higher values of K_d for U(VI) than measured by Honeyman and Santschi, the Honeyman and Santschi data were measured under site-specific conditions. For the purposes of this study, the Honeyman and Santschi conditions were assumed to be representative of Rocky Flats as a whole, and were used to define the upper bound of the K_d distribution for uranium.

For the remaining radionuclides, plutonium and americium, we used the entire range of available data on K_d to define the distribution. The Honeyman and Santschi upper bound for plutonium K_d matches closely with the upper bound reported across the literature, so it was reasonable to use the upper bound reported in the literature. Using the lower bounds identified in the cited literature for all radionuclides allows for the possibility of rapid transport of radionuclides into the groundwater and might help simulate conditions, such as colloidal movement and the special geochemical conditions that promote it, that we are otherwise unable to model.

The distributions were assumed to be lognormal, and the minimum and maximum values described here were assigned to the 0.5% and 99.5% values in the distribution. The properties of a lognormal curve were then used in combination with the two values on the distribution to identify the geometric mean and geometric standard deviation of the distribution. The parameters of the distributions are shown in Table 9.

Table 9. Distributions of K_d Developed for the Independent Calculation (in units of $cm^3\ g^{-1}$ or $L\ kg^{-1}$)

Radionuclide	Geometric mean	Geometric standard deviation
Americium	1800	8.1
Plutonium	2300	5.6
Uranium	2.3	5.4

These distributions of K_d will be used in the independent calculation of soil action levels for Task 5.

It is important to recognize the sensitivity of the K_d value to the aqueous phase concentration and the contaminant transit time. Using ^{239}Pu as an example, we calculated maximum concentration at the receptor well and the time of maximum concentration as a function of the K_d value for a 1 Ci inventory in the source (Figure 1). Maximum concentrations were normalized to the maximum concentration calculated using a K_d value of 2300 mL g^{-1} (1×10^{-9} Ci m^{-3}). The normalized maximum concentration curve ranges over 10 orders of magnitude. The slope of the curve is approximately linear for K_d values less than 1000 mL g^{-1}. For K_d values >1000 mL g^{-1}, the slope increases substantially. The increase in the slope is due to decay effects, because for K_d values >1000 mL g^{-1}, the transit time is greater than the half-life for ^{239}Pu. Also note that for K_d values greater than 100 mL g^{-1}, the time of maximum concentration exceeds 1000 years. Therefore, unless the sampled K_d value is less than 100, groundwater will not be an issue for the scenarios.

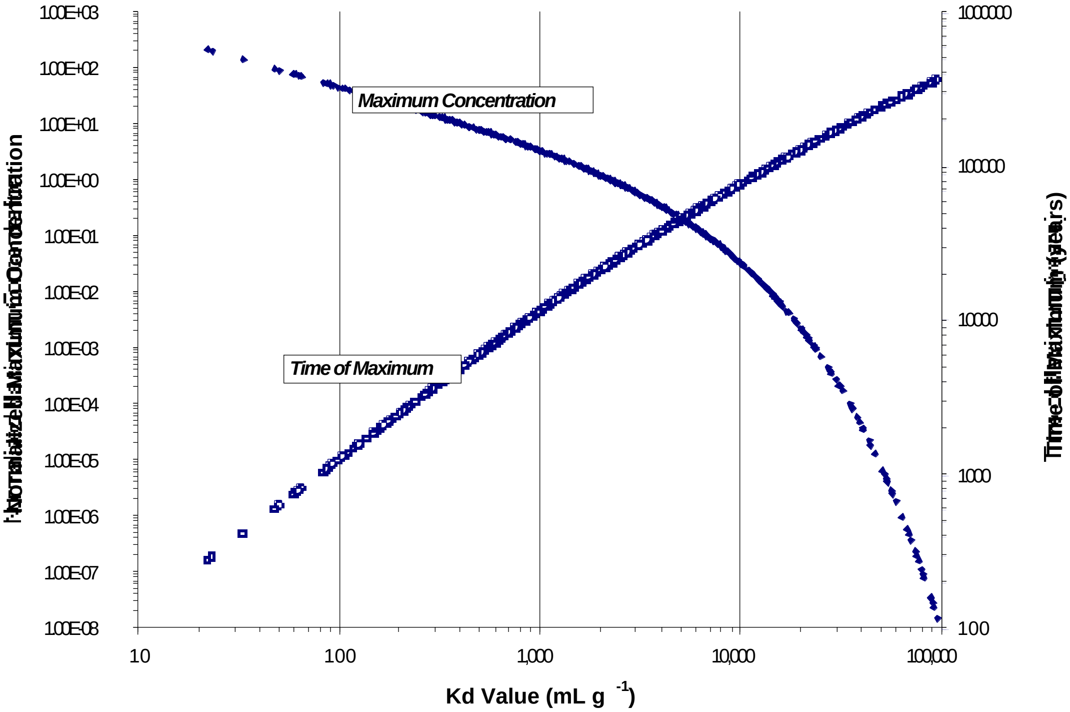

Figure 1. Normalized maximum concentration and time of maximum concentration as a function of the K_d value for ^{239}Pu. Maximum concentrations were normalized to the maximum concentration for a K_d of 2300 mL g^{-1}, which is the geometric mean of the distribution used in the analysis.

We should mention here that the groundwater model employed in RESRAD only considers dissolved phase transport of radionuclides. Recent work by Litaor has suggested that under saturated soil conditions, plutonium can migrate very rapidly. This work is currently unpublished; however, it suggests that certain discrete events, such as heavy rainfall may have moved plutonium into the subsurface in a relatively short period of time. The mechanisms suspected to have resulted in such movement include colloidal transport of plutonium particles through microfractures in the surface soil, and redox reactions coupled with phase changes as a result of saturated conditions near the surface, that temporarily increased the solubility of plutonium. These processes are believed to only operate during periods of heavy rainfall and saturated soil conditions. These processes are still under investigation and are not included in the model, nor can they be given the budget and time constraints of this project. We therefore cannot rule out the possibility that aqueous phase concentrations of plutonium may be underestimated using the approach stated earlier. However, it is our intention to account for the possibility of increased transport conditions such as these by including the lowest measured K_d values available in the literature.

Area of Contaminated Zone

Contamination in soil at Rocky Flats is not uniformly distributed across the site. A number of historic studies have measured concentrations and spatial variation of radionuclides in soil. We used these studies to compile a composite database of soil concentrations at different distances from a significant source of contamination at the site, the 903 Area.

A complication in using RESRAD at Rocky Flats is the highly inhomogeneous spatial distribution of plutonium in the soil. RESRAD works with a specified region of contamination within which the soil concentration is mathematically treated as being uniform, although the developers relax that assumption to accept variation within a factor of 3. Outside the homogeneous region, contamination is assumed to be no greater than background. However, at Rocky Flats, plutonium concentrations in the soil increase by more than a factor of 100 from Indiana Street westward to the 903 Area. Thus, it is difficult to assign a region to a scenario that meets the developers' guidance. If the assigned region is too small, it excludes most of the radioactivity. If it is too large, it fails the test for homogeneity.

To avoid having to conform to RESRAD's definition of contaminated area, we used site data (including air monitoring) to establish relationships between concentrations in air and soil and used these relationships in applying RESRAD to the site. To carry out this task, it was necessary to construct a model of ^{239}Pu concentration in soil as a function of location.

To develop this model, we began with a suitable database of observations. We restricted our selection, for the most part, to measurements for which the documentation included the sampling depth and an approximate time when the samples were taken. One series of measurements that did not meet these criteria is discussed below. The sampling depth is important because recent field and theoretical work reported by Webb et al. (1997) established a fractional concentration depth profile for ^{239}Pu at Rocky Flats that can be applied generically to adjust samples taken at various depths to a common basis.

In general, we followed the example of Webb et al. (1997) and used the ^{239}Pu concentration in the 0–3-cm (0–0.03-m) layer as representative of resuspendable soil and plutonium. The generic profile indicated that essentially all plutonium in the soil at Rocky Flats is currently confined to a depth of 20 cm (0.2 m), with a concentration that decreases with increasing depth. We then adjusted concentrations based on samples taken to depths <20 cm to the 0–3-cm depth by hypothesizing a profile for the sample that was proportional to the standard of Webb et al. (1997). The calculation accounted for plutonium that might have migrated beyond sampling depths less than 20 cm, and a consistent proportion was assigned to the 0–3-cm layer.

Evolution of the depth profile over time is less clear. It appears that after its windborne transport from the 903 Area, plutonium migrated within a few years (at most) into the soil where it was deposited and established the 20-cm profile. Krey and Hardy (1970) indicated that plutonium had already migrated beyond the 13-cm depth. Poet and Martell (1972) questioned this conclusion, reporting that most of the plutonium at seven sites they had sampled was confined to the 0–1-cm layer. They asserted that most of the plutonium found at greater depths in the Krey and Hardy (1970) study occurred at sites that were remote from the 903 Area and in locations where soil had been disturbed. Krey (1974) subsequently defended the conclusion of Krey and Hardy (1970).

Webb (1996) summarized estimates of the soil plutonium inventory from several investigations. These estimates are consistent with a regression curve that shows an initial removal of about 40% of the inventory from the 0–3-cm layer in 10 years (Figure 2). The term "regression" refers to a statistical procedure that fits a function or model, which might be visualized as a curve, to a set of data. The procedure can be extended to use the distances of the data points from the fitted curve (called "residuals") to estimate uncertainties in quantities associated with the model.

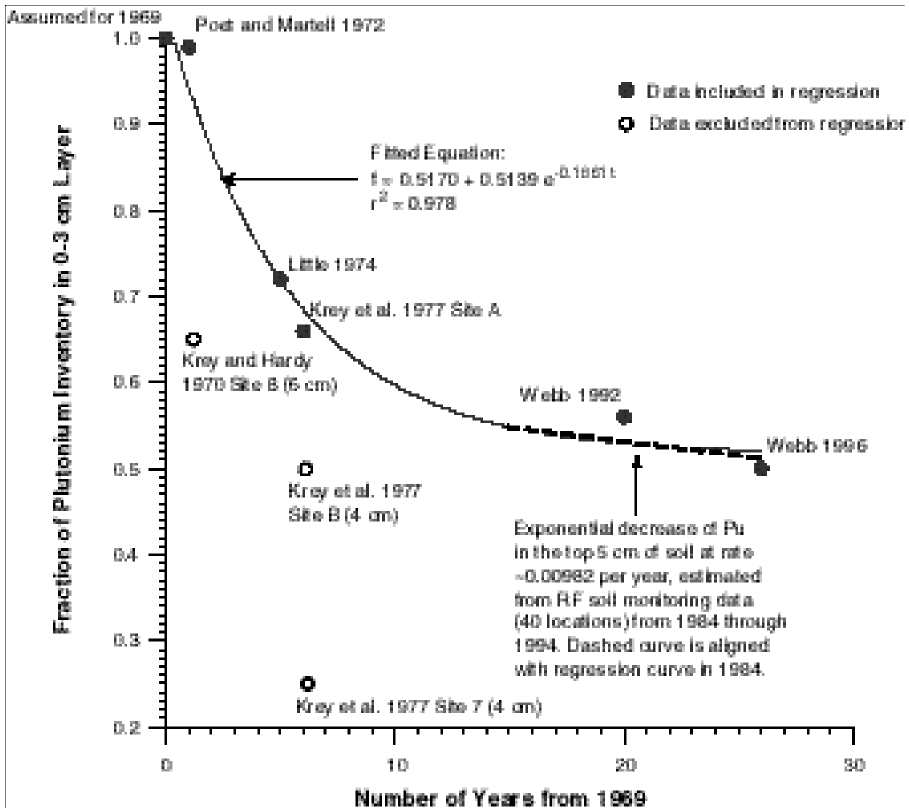

Figure 2. Regression curve fitted to ^{239}Pu data for the 0–3-cm layer of soil at Rocky Flats. The regression was based on data summarized by Webb (1996), which are plotted as black circles. The open circles were excluded from the regression. The dashed line represents an estimated exponential removal rate of plutonium from the 0–5-cm layer measured at 40 stations from 1984 through 1994.

This regression curve, presented in Rood and Grogan (1999), indicates an asymptotic[b] level of about 52% of the initial deposition of plutonium remaining in the 0–3-cm layer. This schedule of decreasing plutonium concentrations is too gradual to be consistent with the conclusions of Krey and Hardy (1970) and with some observations of Krey et al. (1977). Data from some of the locations sampled in these two studies were omitted from the regression because of the apparently inconsistent interpretations. These omitted observations are presented as open circles in Figure 2. Rood and Grogan (1999) has a fuller discussion of the issues involved. The regression curve in

[b] asymptotic refers to the gradual approach of the descending curve to a horizontal line

Figure 2 does not explicitly represent details of the mechanisms of transport in the soil. Rather, the form of the function is based on a simple removal model with partial retention.

It is very likely that natural processes continue to remove plutonium from the surface soil, even though the regression curve suggests that the level would never drop below 50% of the initial deposition. *RAC* performed a statistical analysis on samples from the 0–5-cm depth that were collected as part of the Rocky Flats monitoring program. These data were sampled annually from 1984 through 1994 at 40 locations, with distances roughly 1.6 km (1 mi) and 3.2 km (2 mi) from the center of the site and in all directions at intervals of 18°. Using the aggregated data from these locations, we estimated a loss rate of about 1% per year during the 11-year period. A 95% confidence interval for the rate coefficient is −0.0098 ± 0.0182 (−0.0280, 0.0083) per year. Note that this interval includes a segment of nonnegative numbers and, thus, does not exclude zero loss at the 95% level (however, a 70% confidence interval *would* exclude the zero loss rate). Separate estimates based on the inner and outer circles of sample locations were consistent, giving nearly identical estimates of the rate coefficient.

In assembling the database for the spatial model of plutonium in soil, *RAC* used the fractional concentration profile of Webb et al. (1997) to express concentrations from various depths in terms of the 0–3-cm layer. We have not yet made adjustments to account for the development of the profile over time, but we are studying ways of incorporating this refinement for Task 5.

The raw soil concentration data for ^{239}Pu were obtained from two sources: (1) Table I-2 of Appendix I from Ripple et al. (1994), and (2) a computer archive of 1122 results of soil samples, deposited with the CDPHE by M.I. Litaor. This archive provided Colorado State Plane (CSP) coordinates (in feet) and activity concentrations (in picocuries per gram) for observations reported by Illsley and Hume (1979). It also provided the CSP coordinates for the 40 locations of the Rocky Flats monitoring series mentioned previously (rings at approximately 1.6 and 3.2 km [1 and 2 mi] from the center of the site, at angular intervals of 18°). For each of these 40 locations, we averaged the series ^{239}Pu for 1984–1994 for use in our model; we took the plutonium results for these locations from the 1994 environmental monitoring report (RFETS 1994) rather than from the archive.

Many of the data in the Litaor archive could not be documented and, therefore, were not used. However, one series, with code numbers PT000–PT124, was considered essential because of the coverage that it provided near the 903 Area. The Rocky Flats sampling protocol specified a sampling depth of 0–5 cm, and we assumed that all observations in the PT series were taken in conformity with this protocol. However, it is possible that the series contains some values that are based on shallower depths. We are also uncertain about the dates of sampling for the PT series. It may be possible to obtain further information on the PT series for Task 5. No other data from this archive were used.

The compilation of Ripple et al. (1994) provides good documentation and discussion of a variety of measurements taken during 1969–1971. The protocols vary, and sampling depths range from 1 to 20 cm. The plutonium activity is reported as millicuries per square kilometer, converted to becquerels per kilogram in the database using an assumed average bulk soil density of 1 g cm^{-3}. Coordinates in the appendix of Ripple et al. (1994) were given in the Universal Transverse Mercator (UTM) system (in meters). Litaor's archive included the data from Ripple et al. (1994), which were the basis of what he termed the "historic data set" (Litaor et al. 1995), but this component of the database was taken directly from Ripple et al. (1994). The assembled database from which the *RAC* model is derived consists of 588 entries, and some of the entries represent averages of multiple samples taken at the same location at different times.

Figure 3 shows the locations of all samples in the database. Location symbols are differentiated to indicate concentrations <2, 2–10, 10–100, and >100 Bq kg^{-1} (100 Bq kg^{-1} = 2.7 pCi g^{-1}). Even this crude breakdown gives a fair sense of the spatial distribution of the soil concentrations of ^{239}Pu. Coverage within the plant area and west of the site is relatively thin, and it is unlikely that these areas can be substantially supplemented from other sampling records. Prevailing westerly winds directed most of the attention to areas east of the 903 Area.

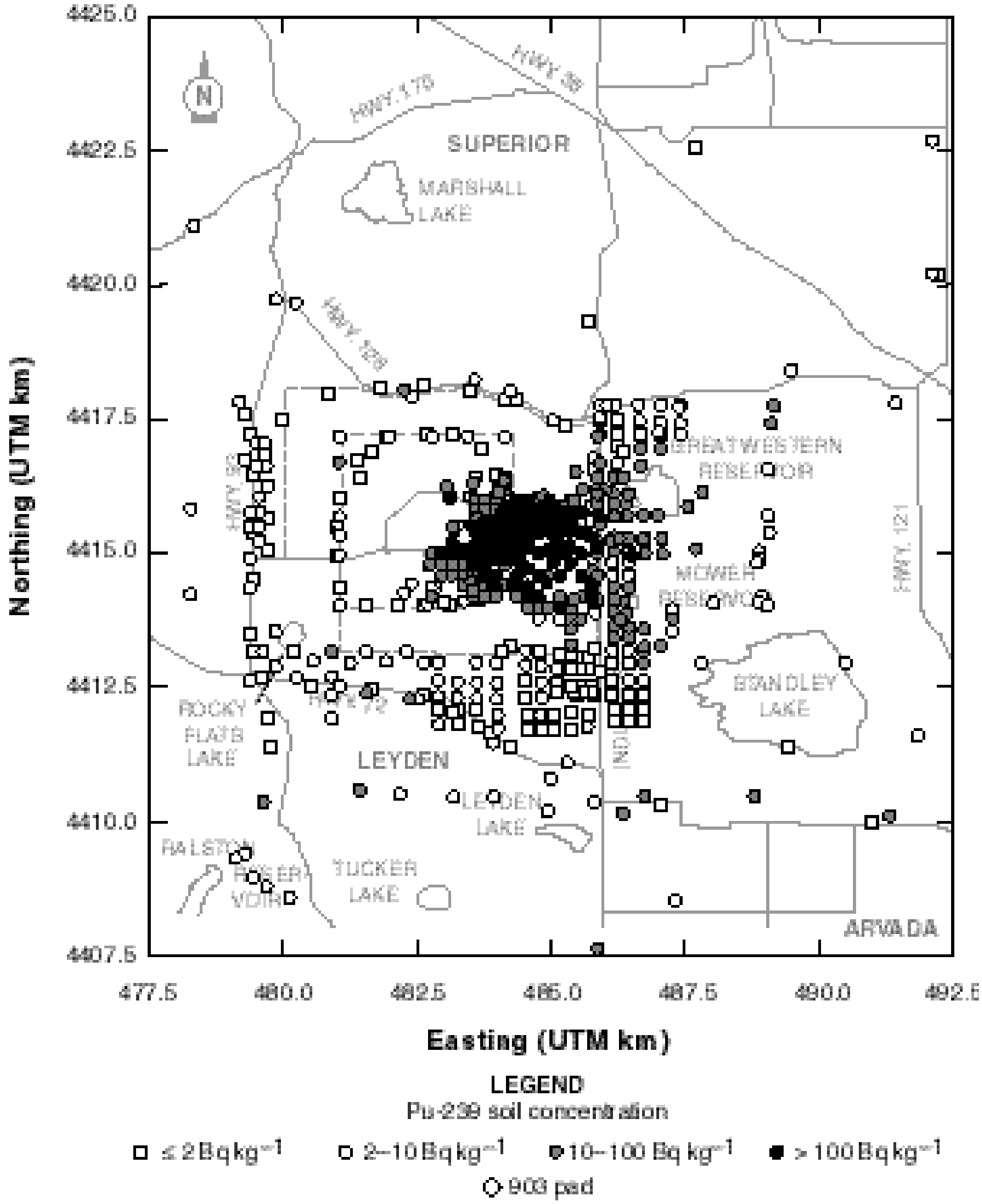

Figure 3. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model (100 Bq kg^{-1} = 2.7 pCi g^{-1}). The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), Ripple et al. (1994), and one series from an archive of M.I. Litaor provided by CDPHE.

To be useful, a spatial model of the plutonium concentration in soil must provide estimates for locations not included in the database by means of interpolation. Also, given the considerable spatial variability in the data, the spatial model must provide smoothing. Some efforts have based estimation of contours on kriging methods (Litaor et al. 1995). The *RAC* approach to smoothing was based on the more direct assumption that most of the spatial signal is the result of wind transport of contaminated soil particles from the 903 Area; therefore, a polar[c] representation from this center is reasonable.

Webb et al. (1997) points out that power functions[d] have given satisfactory fits to data along transects from the 903 Area. Figure 4 shows power functions fitted to subsets of the *RAC* database that lie near the 60°, 90°, and 120° transects; the black squares represent the data of Webb et al. (1997), which we included in our model's database. The Webb et al. (1997) data are extensively documented. Therefore, they provide a check on the transformation of the remaining data from heterogeneous sampling efforts to the common basis represented by the profile given by Webb et al. (1997). This adjusted density profile was also used for soil particles of diameter <2 mm. The 2-mm cutoff corresponds to the sieving separation of rocks from soil used in preparing most of the samples. In some of the older samples, however, the rocks were pulverized and re-mixed with the soil (Krey et al. 1976). Figure 3 shows good consistency of the larger database with the data of Webb et al. (1997), but it also emphasizes the scatter of the data, generally to about a factor of about 10 above and below the curve. If the data corresponding to a temporal evolution of the soil profile is adjusted (Task 5), there may be some change in the fit, but it would be difficult at this time to predict the general effect.

[c] The term "polar" means that we represent any location by its distance from a center (or pole) and the angle that a line drawn from the center to that location forms with a specified direction, usually north.

[d] Power functions have the formula $y = f(x) = Ax^b$, where A and b are constants determined from the curve-fitting procedure (this is an example of regression). In this case, y is the concentration of ^{239}Pu in the soil and x is the distance from the 903 Area. The graph of a power function plotted on logarithmic axes is a straight line. Therefore, when data that are plotted relative to logarithmic axes indicate a straight-line trend, one assumes that they are likely to be satisfactorily represented by a power function.

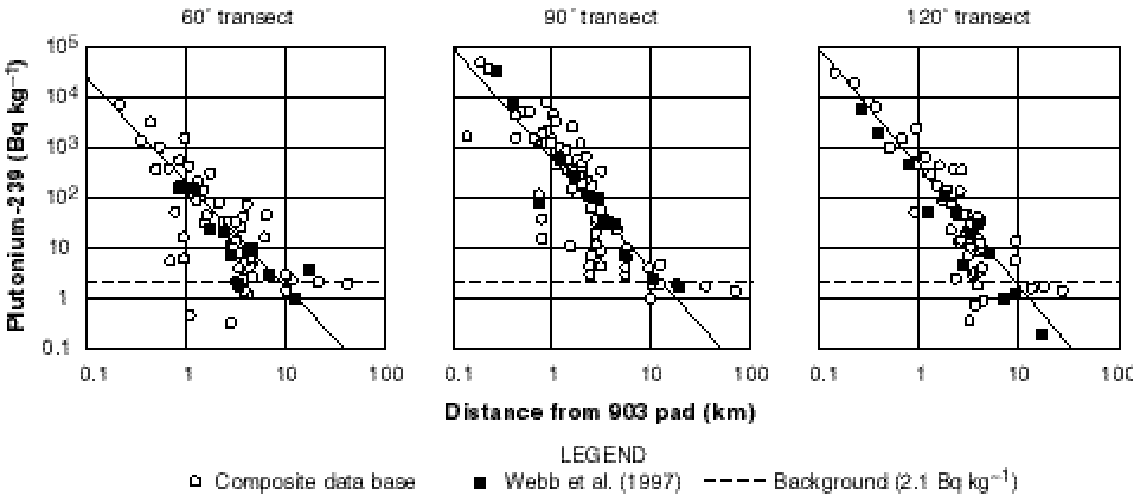

Figure 4. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 Area. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols. Data from all sampling depths have been transformed by the profile of Webb et al. (1997) to represent the 0–3-cm layer.

For the spatial soil model, we fitted a power function to the data within each sector of 22.5°, with centerlines at 0°, 22.5°, 45°, etc. To estimate concentration at points on a sector centerline, the model uses the value of the power function from a point near the 903 Area to the distance at which the power function has the value 2.1 Bq kg^{-1}, which is the estimate of background given by Webb et al. (1997). Beyond this distance, all values are assumed to be background for purposes of the model. Between centerlines of sectors, linear interpolation based on the angle is used to estimate the concentration. For two sectors northwest of the 903 Area (292.5° and 315°), the coverage is inadequate to establish credible power function fits, and the power function for 270° was extrapolated to these two sectors. Contours based on the model are shown in Figure 5.

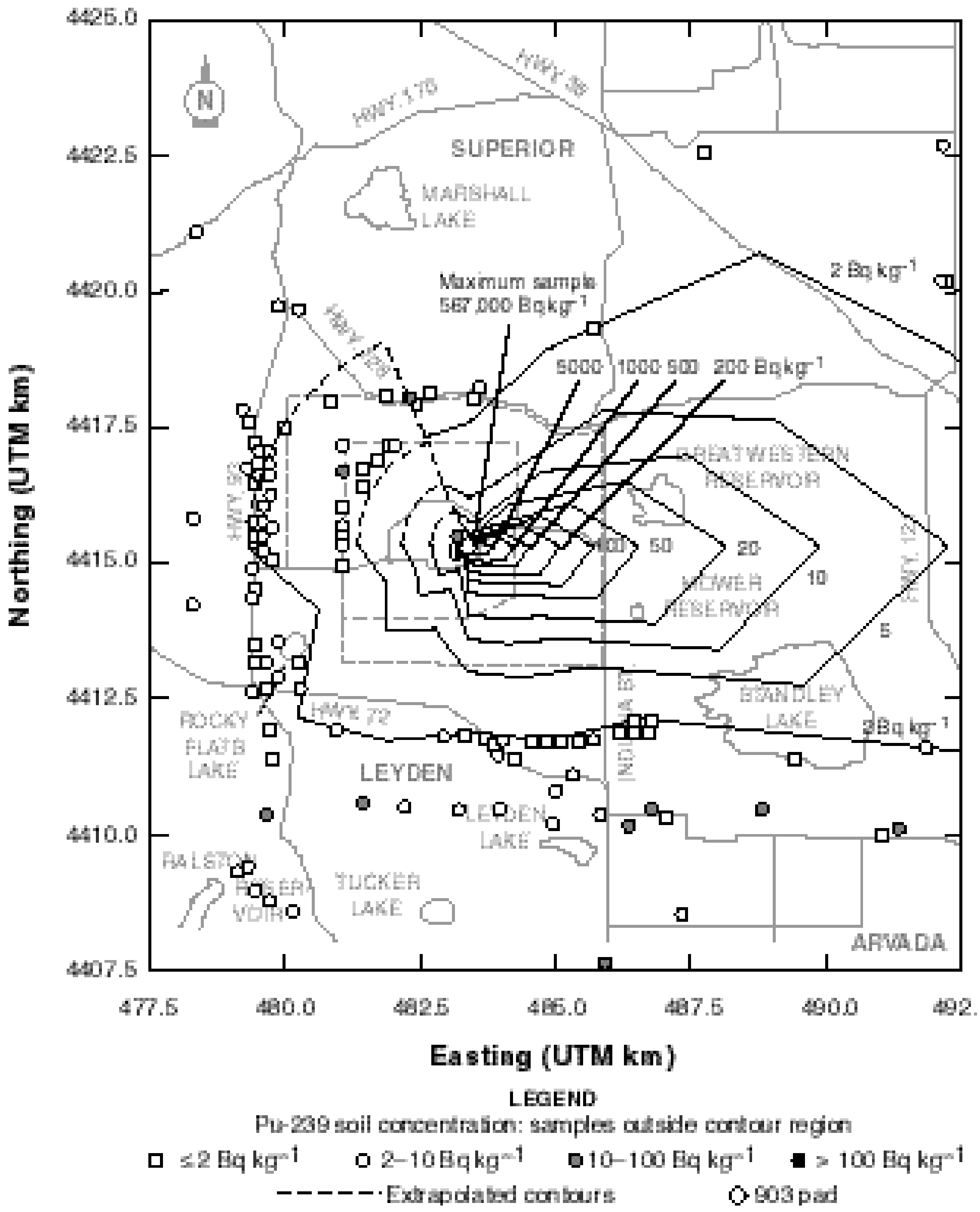

Figure 5. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text.

Dashed lines indicate extrapolation of the two northwest sectors from the 270º sector. Sample locations are shown outside the 2 Bq kg^{-1} (5.4 _ 10^{-2} pCi g^{-1}) contour (approximately background) and within the northwest sectors. Dashed parts of the contours indicate extrapolation where coverage was insufficient for fitting power functions. In these regions and outside the 2 Bq kg^{-1} contour, sample locations were plotted to show that there are some above-background observations where the model would indicate background (2.1 Bq kg^{-1}). However, for purposes of legibility, sample points have been deleted from other regions within the contours.

Although the contours may be considered crude, with an angular resolution no better than the linear interpolation between sectors, they illustrate the considerable variation of the concentrations and the particularly rapid increase as the pad is approached along eastward transects. The model estimates are constrained not to exceed the maximum adjusted sample value (567,000 Bq kg^{-1} or about 15,000 pCi g^{-1}), which occurs in the immediate vicinity of the 903 Area. These contours (or any set of contours based on plutonium concentrations in soil at Rocky Flats) cannot be assumed to provide exact partitions according to magnitude. The smoothing and interpolation provided by the model must be kept in mind. The model is not intended to give accurate estimates at specific locations, but rather it provides a basis for integration[e] of resuspension over large areas for calibration.

Mass Loading Factor

RESRAD bases its calculations of resuspension parameters assuming an area of homogeneous contamination. RESRAD defines an area that has homogeneous contamination as any area where all contamination levels are within a factor of 3 of the mean. An area of higher concentration would then be restricted to an area of no greater than 100 m^2. Figure 5 shows that the soil contamination at Rocky Flats is not homogeneous. Clearly, Rocky Flats soil contamination does not fit within the boundaries of this definition needed for RESRAD.

Because the area of contamination is so closely tied to calculating the resuspension parameter mass loading, we bypassed this calculation in the RESRAD code and had RESRAD estimate resuspension in a different way. The resuspension process, however, is very complex, with a number of mechanisms controlling it that have not been well quantified in spite of the years of research on the topic. Because the best way to evaluate soil resuspension is on a site-specific basis, we calibrated the model to site-specific data.

The RESRAD documentation cautioned that if air concentration values were available for the site under evaluation, these should be used in lieu of the area factor calculation (Chang et al. 1998). There are several sources of air monitoring data across the area of study for the soil action level work. Langer (1991) measured air concentrations at a single location 100 m southeast of the former 903 Area from 1983–1984 and monitored a less instrumented location at the East Gate near the 903 Area. Rocky Flats annual site environmental reports summarize data from several air monitors located throughout the Rocky Flats complex. These monitoring data do not, however, provide particle size information.

The tools that *RAC* used to calibrate resuspension to available air concentration data are described in the Task 5 report. It is important to understand that because of the large degree of inhomogeneity at Rocky Flats, it is difficult to use RESRAD, or most existing assessment

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[e] In this context, “integration” may be thought of as adding up the contributions of resuspension arising from many small areas within the contaminated region to estimate their collective effect on air concentration at a single specified location occupied by an air sampler or the subject of a scenario.

programs, to make these calculations. Our method provides a way to use the RESRAD tool in combination with available site-specific data to make estimates of resuspension based on actual site conditions.

To make this calibration, *RAC* specified the area that was the domain of an individual receptor. Examples might be a ranch for the rancher, some area of land that the recreational user might cover during an exercise period, or the office buildings and surrounding parking area used by the office worker. In general, this area was a small subregion of the contaminated area. We estimated the variation of the air concentration that existed within the defined domain based on the current state of ground cover, using the existing air concentration data. The resuspension mechanism in RESRAD was then constrained to calculate the estimated air concentration for that receptor. This approach bypassed the generic area factor and resuspension mechanism in RESRAD and defined resuspension based on actual site data.

The calibration of the model used a Gaussian plume air dispersion model to predict the annual averaged contribution to plutonium air concentration at a fixed receptor location from resuspension of contaminated soil. The resuspension rate for the calculation was estimated from a soil concentration given by our soil model, meteorological data, and two parameters that need to be estimated for local conditions. For each wind direction, these computed contributions of resuspended material from small areas were added together to provide a total estimate of air concentration. The results were then averaged over the 16 wind directions, using local meteorological frequencies. A prediction was made for the location of each air sampler with trial values for the resuspension parameters, and the results were compared with the monitoring data. This comparison was used to adjust the resuspension parameters to give the best fit of the predictions to the data. The fitted resuspension parameters provided the calibration.

Using these fitted parameters, *RAC* applied the same integration procedure to estimate the annual average of plutonium air concentration at any location on or near the site. We also estimated plutonium air concentrations based on the assumption of reduced soil concentrations that simulate the results of remediation. The regression also yielded estimates of uncertainty for the predicted air concentrations. These air concentrations enabled us to use RESRAD for calculations of dose and soil action levels for any scenario. Anspaugh et al. (1975) described a similar procedure for estimating resuspension rates using data for plutonium from the Nevada Test Site.

A procedure such as this was required because air concentrations within the domain of a scenario depend not only on soil contamination within that domain, but also on soil contamination throughout a larger upwind region. The extent of this larger region is not well defined.

Krey et al. (1976) reported results of soil and air sampling east of the 903 Area. Their comparison of plutonium activity per gram of airborne dust and plutonium activity per gram of soil led them to the conclusion that only 2.5% of the airborne dust was representative of the soil at the three sites they sampled. The remainder of the airborne dust presumably came from outside the immediate vicinity. An uncharacteristic frequency of rain reported by Krey et al. (1976) during their field work suggests some caution regarding the 2.5% figure.

Table 4.1 of NCRP Report No. 129 (NCRP 1999), however, indicated that 95% of the airborne dust at about a 1-m height comes from an upwind fetch (upwind distance) of 60 m if the ground cover is tall grass (145 m for short grass and 175 m for bare ground). These distances seem too short to be consistent with the observation of Krey et al. (1976). Our calculations suggested that at the locations sampled at Rocky Flats, most of the resuspended dust would have come from onsite. There is literature on the subject of footprints of fluxes (the footprint is the source region for a flux through a specified area, such as a sampler intake). Our method implicitly deals with the question by integrating over a large area that is certain to contain the relevant footprint.

It is important to understand the dependence of this calibration on the current state of ground cover. All the available air monitoring data reflect this ground cover; therefore, any calibration done to these data necessarily includes this assumption. *RAC* has developed resuspension parameters for an extreme situation, such as a fire or other natural disaster, that might remove the grass cover and leave an open soil source available for resuspension.

This calibration was developed as a part of Task 5, *Independent Calculation*. We believe this method will make the best use of RESRAD within its design limits and provide external data for quantities that exceed the design limits for this site. RESRAD is well suited to performing radiological decay chain calculations, concentrations of radionuclides in exposure media (given the concentrations in air that our auxiliary calculation will provide), and annual dose at various future times from multimedia exposure to the radionuclides. The corresponding soil action levels for each scenario depend on the highest plutonium soil concentrations that are consistent with the limiting annual dose for the scenario.

Mean Annual Wind Speed

Mean annual wind speed was not required in the previous version of RESRAD. It is used to calculate the area factor for use in the resuspension calculation in RESRAD Version 5.82. According to the National Climatic Data Center (www.ncdc.noaa.gov), the 43-year annual average wind speed for the Denver area is 4 m s^{-1} (NCDC 1999). This average fluctuates very little for any given year, ranging from about 3.7 to 4.4 m s^{-1}.

As described above, however, *RAC* estimated resuspension by calibrating to site-specific air concentration data. This calibration required the use of wind speed data, but it also used a data set that contains more information than wind speed alone. For example, data on wind direction and atmospheric stability class from the onsite Rocky Flats meteorological station were also included. Further information on the use of the wind speed data will be included in the Task 5 report. The joint frequency tables showing the 5-year average wind speed data for the Rocky Flats area are given in Appendix A to this report. The first six tables in the appendix represent the data for each stability class, with fractional values adding up to 1 for each table. The last table is the composite joint frequency table for all stability classes.

Because there was a recognized potential for high wind events at Rocky Flats, we carefully considered them for this project. During Phase II of the Historical Public Exposure Studies on Rocky Flats, a series of high wind events was predicted to result in a significant quantity of offsite contamination from the 903 Area (Weber et al. 1999). It was demonstrated that these winds resuspended a large amount of the available plutonium from the highly contaminated area.

The largest wind events during the period after the 903 Area barrels were cleared and before the area was covered with asphalt were modeled as six discrete wind events. These events produced the largest degree of dust and contamination suspension from the 903 Area. The high wind events were estimated to have been responsible for most of the activity released from the

903 Area. However, high wind speeds also result in greater dispersion, dilution, and depletion within an airborne plume, resulting in lower air concentrations than would be predicted had the same activity been released over a longer period of time and modeled using annual average meteorological data. This is clear if we consider the plutonium concentrations predicted and reported during the Phase I of the Historical Public Exposures Study on Rocky Flats. Although the source term for respirable particles calculated in Phase I was about the same as that calculated in Phase II, the total integrated concentration value from the Phase I work is a factor of 2 to 3 higher than that from the Phase II work. Consequently, it appears that while the discrete events may have contributed to most of the offsite contamination, they do not appear to be as important from an airborne concentration standpoint.

This is a very important characteristic of high winds. Although at the beginning of the Historical Public Exposures Studies on Rocky Flats, high winds were widely regarded as probably the single greatest contributor to exposure, they were revealed in that study to be responsible instead for reducing the concentrations of contamination in air. As a result, high winds will not be explored further in the soil action level project.

SCENARIOS

In the Task 2 report, we described and defined exposure scenarios and explained how they are an integral part of the soil action level work. The goal of establishing radionuclide soil action levels is to protect people who may, in the near or distant future, come into contact with a site where radionuclides contaminate the soil at levels above background. Exposure scenarios describe the characteristics and behaviors of these hypothetical individuals. The people described by the scenarios live, work, or use the Rocky Flats site for recreational purposes.

A goal for designing the scenarios in this study was that if the hypothetical individuals were protected by specified dose limits, then it was reasonable to assume that others would be protected. We have given careful consideration to offsite exposures and have designed the scenarios so that if the person living onsite full-time were protected, then the person living offsite would also be protected. In much the same way as the Clean Water Act defines water intake in defining the levels of allowable contamination, the scenarios are reference standards against which levels of radionuclides in the soil at the Rocky Flats site can be measured.

The scenarios also incorporated physiological characteristics that would affect the estimate of radiation dose that these hypothetical people would receive. Behavioral characteristics are plausible and relevant to the exposure situations and the radiation protection objectives. Because this study is prospective and has the goal of protecting potentially exposed people from radiation in the future, it was necessary to consider several exposure scenarios to cover the varied and possible uses of the land in the future.

Scenarios were not included in the sensitivity analysis and each scenario parameter is treated deterministically in our calculations. Each scenario hypothesized the exposure characteristics of a single individual, and that individual has a defined set of characteristics. We are not representing a population of people with scenarios but rather defining possible lifestyles that an individual might have in the future at Rocky Flats.

RAC evaluated the three scenarios described in the existing soil action level report (DOE/EPA/CDPHE 1996), along with four additional scenarios that we proposed after numerous discussions with the RSALOP at the monthly soil action level meetings. *RAC* designed specific scenarios during the months of discussion with the Panel and added others at the request and suggestions of the Panel. We initially considered ten proposed scenarios, and these scenarios were briefly described in the Task 2 report. As discussions continued, *RAC* recommended and the Panel agreed that some of the proposed scenarios were very similar to the three DOE/EPA/CDPHE scenarios described in the Rocky Flats Cleanup Agreement. We considered the three DOE/EPA/CDPHE scenarios as plausible scenarios for the Radionuclide Soil Action Level project.

Table 10 lists the seven scenarios that are currently being evaluated, with the *RAC* scenarios grouped as nonrestrictive and restrictive. Nonrestrictive means that the hypothetical individual has no restriction to the site in terms of time or location. The restrictive scenarios mean that the person's time or location is limited while on the Rocky Flats area. Because the future land use cannot be known with certainty, it was important to include both types of scenarios for evaluation.

Table 10. Summary of Final Scenarios for Evaluation

	RAC scenarios	
DOE/EPA/CDPHE scenarios	Nonrestrictive (full-time)	Restrictive (part-time)
Residential	Rancher	Current onsite industrial worker
Open space user	Infant of rancher	
Office worker	Child of rancher	

The Rocky Flats Cleanup Agreement scenarios have been described previously (DOE/EPA/CDPHE 1996). The additional *RAC* scenarios include

1. Nonrestrictive (full-time): The resident rancher scenario assumes future loss of institutional control. The rancher is raising a family, maintaining a garden, and leading an active life at the site, spending 24 hours per day, 365 days per year, or 8760 hours at the site. Of that time, over 40% of the ranchers time is spent outdoors. The potential pathways of exposure for this person include inhalation, eating produce from a garden irrigated with water from a well (groundwater), direct soil ingestion from outdoor activities, ingestion of drinking water from an onsite well (groundwater), and direct gamma exposure from the soils and airborne radioactivity. The annual breathing rate is 10,800 m^3 per year, based on a time-weighted average of breathing rates and activity levels.
2. Nonrestrictive (full-time): The child of the rancher family is assumed to be a 10 year old and onsite 24 hours per day, 365 days per year, or 8760 hours per year. The potential pathways of exposure include inhalation, eating produce from a garden irrigated with water from an onsite well (groundwater), ingestion of drinking water from an onsite well (groundwater), direct soil ingestion, and gamma exposure from soils and airborne radioactivity.
3. Nonrestrictive (full-time): The infant in rancher family is 2 years of age and onsite 24 hours per day, 365 days per year, or 8760 hours per year. The infant's potential pathways of exposure include inhalation, eating produce from a garden irrigated with water from an onsite well (groundwater), ingestion of drinking water from an onsite well (groundwater), some direct soil ingestion from outdoor activities, and direct gamma exposure from soils and airborne radioactivity.
4. Restrictive (part-time): The current onsite industrial worker scenario assumes a person works onsite 8_ hours per day, 5 days per week, 50 weeks a year, or 2100 hours per year. It is assumed that 60% of the worker's time is spent outdoors. The potential pathways of exposure for this person include inhalation, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils. The annual breathing rate is 3700 m^3 per year, based on a time-weighted average of breathing rates and activity levels for the time spent onsite.

It is important to remember the difference between restrictive and nonrestrictive when reviewing the scenario characteristics. The *RAC* restrictive scenario assumed 60% of the current onsite industrial worker's time spent onsite is outdoors, for a total of approximately 1200 h y^{-1}

outdoors. The rancher, on the other hand, appears at first glance to spend less time outdoors, only 40% of the time. But with a total time onsite of 8760 h y^{-1}, the rancher spends approximately 3500 of those hours outdoors. Each person represented by a scenario is present at the site for a defined period of time.

For the soil action level assessment, the scenarios are described and defined by numerous parameters, some much more important than others. The scenario parameters include breathing rates for various activity levels and ages, soil ingestion rates for children and adults, fraction of time spent indoors and outdoors, and the potential use of or exposure to contaminated water from the area. We have focused our greatest effort on establishing values for breathing rate and soil ingestion, as these are parameters in which the Panel expressed primary interest. For the remaining parameters, we used the literature to select values, which in some cases differ from the RESRAD default values or the DOE/EPA/CDPHE scenarios (DOE/EPA/CDPHE 1996). Table 11 summarizes the key parameter values for all scenarios.

Table 11. Scenario Parameter Values for DOE and *RAC* Scenarios

	DOE/EPA/CDPHE scenarios			*RAC* recommended scenarios			
				Nonrestrictive			Restrictive
Parameter	Residential	Open space	Office worker	Resident rancher	Child of rancher (10 y)	Infant of rancher (2 y)	Current site industrial worker
Scenario name	DOE-1	DOE-2	DOE-3	*RAC*-1	*RAC*-2	*RAC*-3	*RAC*-4
Dose limit (mrem y^{-1})	15/85	85	85	15	15	15	85
Onsite location				East of present 903 Area	East of present 903 Area	East of present 903 Area	Present industrial area
Time on the site (h d^{-1})				24	24	24	8.5
Time on the site (d y^{-1})				365	365	365	250
Time on the site (h y^{-1})	8400	125	2000	8760	8760	8760	2100
Time indoors onsite (h y^{-1})				5300	6600	7740	900
Time indoors onsite (%)	100	100	100	60	75	90	40
Time outdoors onsite (h y^{-1})	0	0	0	3500	2100	860	1200
Time outdoors onsite (%)	0	0	0	40	25	10	60
Breathing rate (m^3 y^{-1})	7000	175	1660	10800	8600	1900	3700
Soil ingestion (g y^{-1})	70	2.5	12.5	75	75	75	50
Irrigation water source	Ground-water	NA[a]	NA	Ground-water	Ground-water	Ground-water	NA
Irrigation rate (m y^{-1})	1	NA	NA	1	1	1	NA
Onsite drinking water source	no	no	no	Ground-water	Ground-water	Ground-water	no
Drinking water ingestion (L d^{-1})	NA	NA	NA	2	1.5	1	NA
Drinking water ingestion (L y^{-1})	NA	NA	NA	730	550	365	NA
Fraction of contaminated homegrown produce	1	0	0	1	1	1	0
Fruits, vegetables and and grain consumption (kg y^{-1})	40.1	NA	NA	190	240	200	NA
Meat (kg y^{-1})	NA	NA	NA	95	60	35	NA
Milk (L y^{-1})	NA	NA	NA	110	200	170	NA
Leafy vegetables (kg y^{-1})	2.6	NA	NA	64	42	26	NA

[a] NA = not applicable.

To select appropriate parameters for the scenarios, we reviewed the scientific literature and current EPA and NCRP guidance. For two of the parameters that are particularly important in the scenarios (breathing rate and soil ingestion rate), we fully considered the uncertainty (or variability) distributions of these parameters. For these two parameters, we generated a distribution of values and sampled from the distribution using Monte Carlo techniques. This process considered the available studies equally. The distributions are characterized with a central value, such as the median, and some measure of the spread of the distribution, such as the standard deviation or the 5th and 95th percentiles of the distribution.

In developing a particular scenario and considering variability of a parameter within the population studied, we selected a percentile of the distribution as needed to extend protection to a larger fraction of a potentially exposed population with characteristics similar to those of the scenario subject. After the parameter value was selected from our distribution of values for use in the scenario, the scenario was considered fixed just as standards are fixed as a benchmark against which to measure an uncertain value.

The following sections provide details on selecting the scenario parameters that are expanded or differ from the parameter values given for the current DOE/EPA/CDPHE scenarios.

Breathing Rate

We compiled data from numerous published papers to provide perspective in selecting suitable breathing rates (Table 12). In general, breathing rate studies indicate that gender makes little difference on breathing rates through about age 12. For teens through adulthood, the breathing rate can be 40–50% higher in males than females. There is also age dependency on breathing rates, with adults having breathing rates that are about a factor of 3 higher than for young children. For a person of a given age and gender, the most significant parameter affecting breathing rate is the level of activity; breathing rates can be 15 times higher under maximum work conditions than resting. This activity dependence is important for acute exposure of a few hours, but less important for a continuous chronic exposure of a year.

The time for each *RAC* scenario was divided among three types of activities: sleeping or sedentary, light activity, and heavy activity. For the infant and child, the time was divided into sleeping and light and moderate activities. For the onsite worker, the time was divided between time at the site (hours per day) and time away from the site (hours per day). While at the site, the time spent in light, moderate, and heavy activity was identified. For each scenario, we then assigned duration for the various daily activity levels. The daily breathing rate for each scenario was the time-weighted average breathing rate for each activity level. Although there is no distinction between indoor and outdoor air concentrations in the assessment, the levels for indoor and outdoor activities differed.

Based on published studies, *RAC* created distributions of breathing rates for active and sedentary adults, for active and sedentary children, and for active and sedentary infants. Using these distributions and the recommended breakdowns of daily activity for each receptor, we created distributions of scenario breathing rates for each scenario. *RAC* recommended and the Panel agreed to using the 95th percentile value from these distributions for the scenario breathing rate. Figure 6 shows the distributions for the nonrestrictive scenarios (rancher, child, and infant), and Figure 7 shows the probability distribution for breathing rates for the restrictive scenario (onsite worker).

Table 12. Summary of Key Breathing Rate Studies Reviewed

Study	Approach	Group	Breathing rate ($L\ min^{-1}$)
Silverman et al. (1951)	Max inspiration and expiration determined for design of respiratory equipment; one group; adult males	Male athlete, sitting on bicycle	10.2
		heavy exercise	75
Thompson and Robison (1983)	Based on breathing rate at normal body temperature and pressure; nine age groups; infant through adult; male and female	Adult male resting	8.8
		active	30
Roy and Courtay (1991)	Based on time budgets (hours spent at various activities); six age groups; infant through adult; male and female	Adult male resting	7.5
		Adult male, heavy activity	50
Layton (1993)	Based on oxygen uptake associated with energy expenditures and metabolism; seven age groups; infants through adult; male and female	Adult male average	11
		Range based on activity during day	7–12
Finley et al. (1994)	Age-specific distributions for chronic inhalation rates based on Layton (1993)	Adults 50th percentile	8.2
		Adults 5th, 95th percentiles	5.8, 11.6
EPA (1997)	Deterministic; Outdoor workers (15 men; 5 women)	Light activity	12
		Heavy activity	51
	Outdoor construction workers (19 males)	Light activity	24
		Heavy activity	34

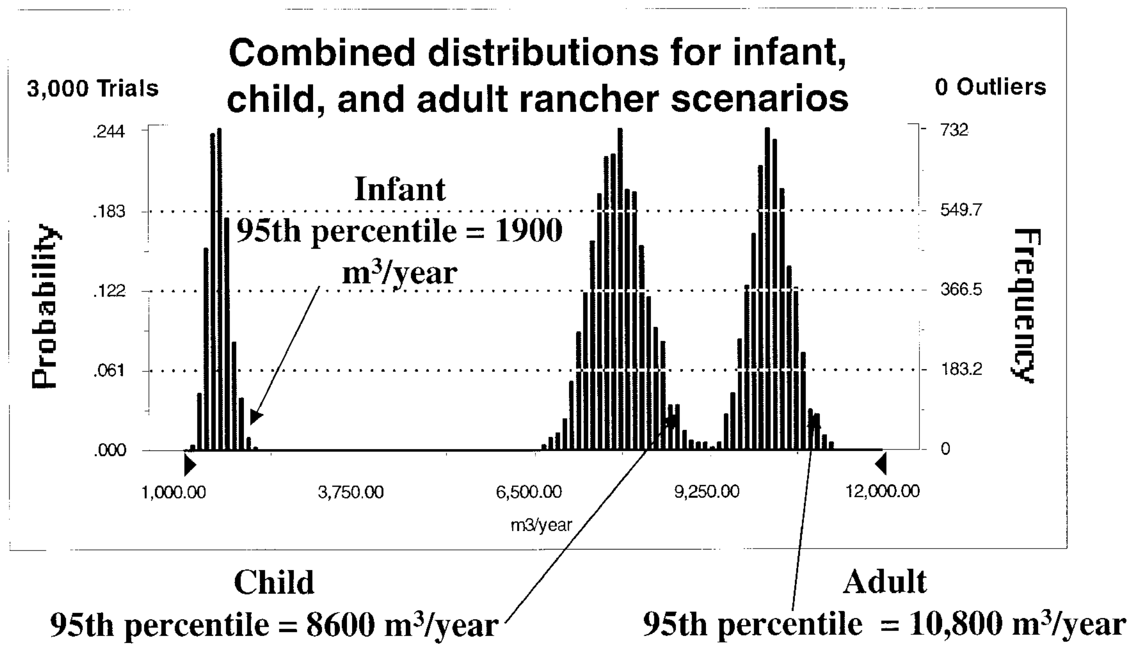

Figure 6. Distributions of breathing rates for the nonrestrictive scenarios: infant, child, and rancher. The 95th percentile of the distribution is shown for each scenario.

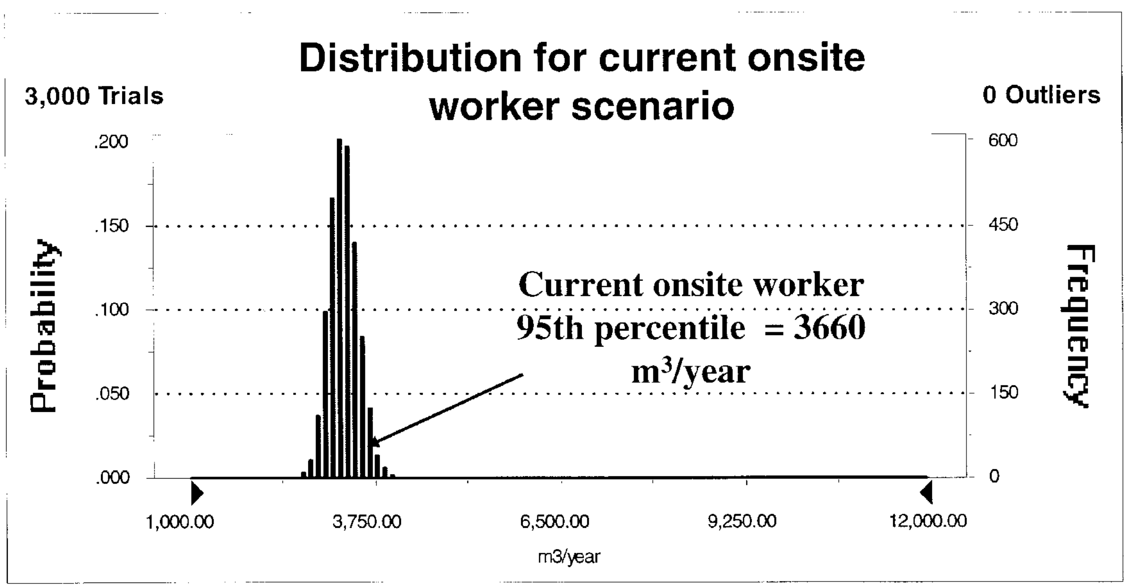

Figure 7. Probability distribution of breathing rate values for the restrictive scenario: current onsite industrial worker scenario. The 95th percentile of the distribution is 3660 $m^3\ y^{-1}$.

Soil Ingestion

Various studies have evaluated the unintentional and intentional ingestion of soil by children and adults. Table 13 lists the studies used in selecting the soil ingestion rate for our scenarios. The table summarizes the approach used in assessing ingestion in each study and the geometric mean and geometric standard deviation for those studies. In 1984, the Centers for Disease Control estimated age-specific soil ingestion at about 10 g d^{-1} based on observations of behaviors of children of 1 to 4 years of age (Kimbrough et al. 1984). In 1986, one of the first quantitative assessments of human soil ingestion was carried out using tracer elements in the soil like aluminum, silicon, titanium (Binder et al. 1986). In 1990, Calabrese et al. (1990) studied soil ingestion rates in adults and children using a mass balance approach and more controlled procedures. Simon (1998) developed scenarios based on an extensive review of the literature. The scenarios applicable to this current soil action level study are for a rural lifestyle with homes in a sparsely vegetated area, similar to the Rocky Flats area. Simon assumed a lognormal distribution for inadvertent soil ingestion for adults with a geometric mean of 0.2 g d^{-1} and a geometric standard deviation of 3.2. For children living this lifestyle, the geometric mean is 0.2 g d^{-1}, with a geometric standard deviation of 4.2 to develop a distribution of values, and a median estimate of 0.2 (which would give 5th and 95th percentile values of 0.02 g d^{-1} and 2 g d^{-1}, respectively).

Soil ingestion is difficult to verify and quantify, and some studies do not differentiate between inadvertent or intentional intake. Both inadvertent and intentional soil consumption is seen worldwide, in all cultures, and intentional soil consumption can affect estimates of soil ingestion rates selected for use in this prospective study. During our discussions with the RSALOP, questions arose regarding soil ingestion values and how the extreme behavior of geophasia (intentionally consuming soil) might affect our probability distribution. There was concern that the few geophasic individuals in some of these studies biased our initial use of the 95th percentile value for daily soil ingestion rate extremely high. Many soil ingestion studies have focused primarily on children, leading to a general view that geophasia is more common in young children than other segments of the population. The reason for this conclusion may be that it has been easier to document geophasic children in the more controlled study environments with children. However, there are several studies (e.g., Simon 1998) that cite cases of geophasia in several segments of the population, including adolescents and pregnant women. While this may be more common in indigenous or rural populations, geophasia has been documented in various population subgroups in United States. The incidence of geophasia in the population is quite small, estimated at less than 1%; however, quantitative evaluation of this phenomenon is sparse.

Most studies, even the more recent mass-balance soil ingestion studies (Stanek and Calabrese 1995) are conducted under fairly idealized conditions or during more mild seasons of the year, and authors tend to point this out in their reports (Calabrese et al. 1990; Binder et al. 1986). This timing factor provides conditions where children may have more ready access to open play areas and outdoor activities and adults are more involved in gardening activities. While values derived from studies conducted from a few days to a few weeks are quite valid in estimating daily soil ingestion rates, there is a need to carefully consider the implications of translating this daily soil ingestion rate to an annual soil ingestion rate.

Table 13. Summary of Soil Ingestion Studies Reviewed

Study	Approach	Soil ingestion ($g\ d^{-1}$) Geometric mean	Geometric std dev
Simon (1998)	Scenarios based on literature review:		
NCRP Report 129	Rural lifestyle (w/homes)—sparsely vegetated		
(NCRP 1999)	Lognormal adults (inadvertent)	0.2	3.2
	Lognormal children (inadvertent)	0.2	4.2
Thompson and Burmaster (1991) (reanalysis of Binder et al. 1986)	Lognormal distribution (children)	0.06	2.8
Stanek and Calabrese (1995) (reanalysis of Calabrese et al. 1990)	Range of median soil ingestion of 64 children over 365 days	0.001–0.10	
	Median of daily average soil ingestion of 64 children:	0.075	
	Range of upper 95% soil ingestion estimates	0.001–5.3	
	Median upper 95% soil ingestion estimate of 64 children over 365 days	0.25	
Calabrese et al. (1990) (children)	Distribution percentiles Median (5th, 95th percentiles)	0.02 (0, 1.2)	
Thompson and Burmaster (1991) (included geophasic children)	Distribution percentiles Median (5th, 95th percentiles)	0.06 (0.01, 9)	
Kimbrough et al. (1984) (children)	Deterministic Mean (low, high)	0.1 (0.05–5)	
Hawley (1985) (adults)	Deterministic (average estimate)	0.06–0.07	
EPA (1997) (adults)	Deterministic (conservative)	0.1	
NCRP Report 123 (NCRP 1996)	Deterministic (conservative)	0.25	

The daily soil ingestion rates are based on a few days or weeks of measurements during times when the soil ingestion may be more likely because of weather conditions or available surface soil. When converting this rate to an annual intake, care must be given because the year includes large periods of time where outdoor inadvertent soil ingestion activities may be somewhat limited by snow cover, frozen ground, and inclement weather. For these reasons, we will use the 50th percentile of our distribution for our daily soil ingestion rate. From the daily soil ingestion rate, we will then calculate an annual soil ingestion value based on the number of days of exposure.

We reviewed various published soil ingestion studies and fit a probability distribution to the data from these studies (NCRP 1999; Simon 1998; Stanek and Calabrese 1995; Thompson and Burmaster 1991; Calabrese et al. 1990). We then looked at how deterministic values from other studies fit into the probability distribution (Kimbrough et al. 1984; EPA 1997; NCRP 1996; Hawley 1985). Figure 8 shows the probability distribution for the soil ingestion studies. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown. As stated above, we used the 50th percentile of this distribution (0.2 g d^{-1}) as the daily soil ingestion rate for our scenarios.

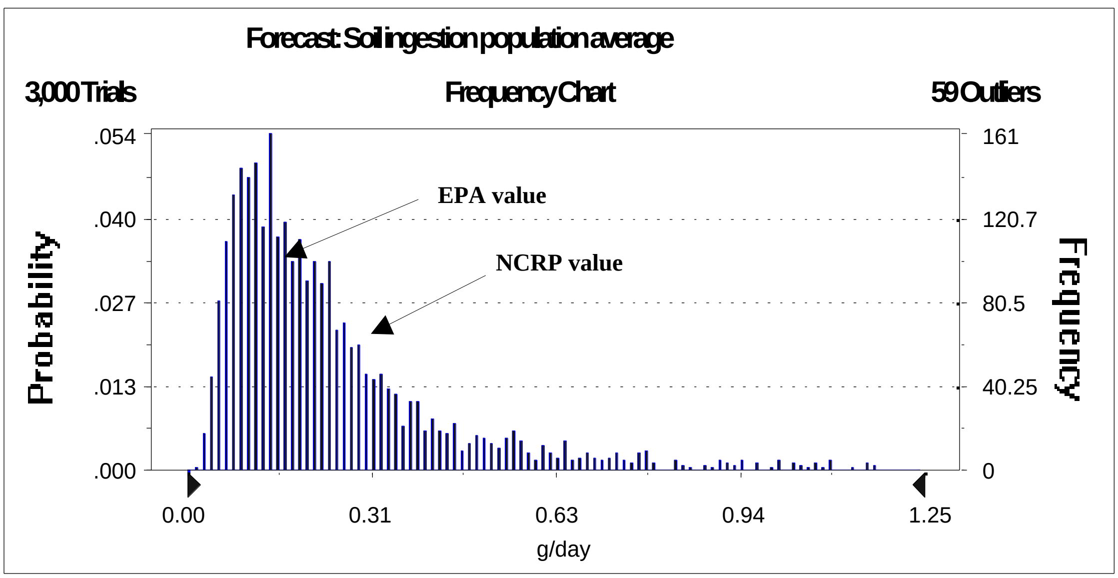

Figure 8. Frequency distribution of soil ingestion values from CrystalBall©. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown.

Groundwater as Irrigation and Drinking Water Source

While groundwater was a source of drinking water and irrigation for the rancher scenario, it has been emphasized that no elaborate calculations can be undertaken for this pathway within the scope of this project. The effort will be restricted to the models and mechanisms that are incorporated within the codes under consideration, with all relevant caution. The irrigation fraction from groundwater for the rancher scenario was 1.0, the RESRAD default value. The contamination fractions of drinking water and irrigation water for the rancher scenario were both 1.0, the default parameter values for RESRAD.

As discussed in the Task 2 report for this project (Killough et al. 1999), the DOE/EPA/CDPHE scenarios (DOE/EPA/CDPHE 1996) did not include the groundwater and surface water pathways because (1) the site streams (Woman and Walnut Creeks) are perennial and would not provide a reliable year-round water source for an individual living on the site and (2) surface aquifers underlying the site do not produce enough water for domestic or agricultural use. The aquatic food pathway was eliminated because the streams are not capable of sustaining a viable fish population. We have reviewed the DOE/EPA/CDPHE approach and agree with their conclusions with regard to surface water pathways. Regarding the groundwater pathway, however, it is not unreasonable to assume for the rancher scenario living under subsistence conditions, a water well that produces 2 gal min^{-1} (DOE 1995b) would be adequate to provide drinking water and perhaps water for a few head of livestock and some limited irrigation. By addressing these pathways, even on a screening level, we can evaluate their potential importance.

Drinking Water Intake

We recommended a drinking water intake of 2 L d^{-1} (730 L y^{-1}) for the adult rancher scenario, 1.4 L d^{-1} (550 L y^{-1}) for the child of the rancher, and 1 L d^{-1} (365 L y^{-1}) for the infant of the adult rancher. These values are based on regulatory guidance from the EPA (1989, 1997) and from other studies (Finley et al. 1994). The current DOE/EPA/CDPHE scenarios did not include drinking water as a potential pathway. The RESRAD default value for drinking water ingestion is 510 L y^{-1}.

Fruits, Vegetables, and Grain Consumption

Annual consumption of major food groups as a function of age for the United States have been estimated and reported by various agencies. This information was necessary in our assessment in order to calculate an average dose from ingestion of produce and grains grown in the contaminated soil, or of meat and milk ingested from animals that ate vegetation grown on the site. In a recent publication, NCRP (1999) compiled values from a number of sources for consumption of major food groups. We recommended an annual consumption rate for fruits, nonleafy vegetables, and grains of 190 kg y^{-1} for the rancher scenario, 240 kg y^{-1} for the child scenario, and 200 kg y^{-1} for the infant scenario (Table 5.1, NCRP 1999). Consumption of leafy vegetables is assessed separately in RESRAD. For the *RAC* scenarios we assumed the consumption of leafy vegetables at the rate of 64 kg y^{-1} for the rancher, 42 kg y^{-1} for the child, and 26 kg y^{-1} for the infant scenarios (Table 5, NCRP 1999). The DOE/EPA/CDPHE scenarios assumed 40.1 kg y^{-1} of vegetables, fruits, and grains and 2.6 kg y^{-1} of leafy vegetables. The

RESRAD default values for these parameters are 160 kg y^{-1} for fruits, nonleafy vegetables, and grains and 14 kg y^{-1} for leafy vegetables.

Milk and Meat Consumption

We recommended an annual ingestion rate for milk of 110 L y^{-1} for the adult rancher, 200 L y^{-1} for the child, and 170 L y^{-1} for the infant (NCRP 1999). For meat consumption, we recommended a value of 95 kg y^{-1} for the adult rancher, 60 kg y^{-1} for the child, and 35 kg y^{-1} for the infant (NCRP 1999). These pathways were not assessed for the DOE/EPA/CDPHE calculation.

CONCLUSIONS

To develop meaningful and appropriate calculations of soil action levels at Rocky Flats, *RAC* collected site-specific data and presented them in this report. Data of this type will be used for all parameters that were revealed as sensitive to change and parameters that warranted adaptation based on the information available in the literature. Not every parameter necessary for the use of RESRAD was changed from its value in the original set of calculations (DOE/EPA/CHPHE 1996). Changes were often not necessary because the values were not sensitive to change, and effort expended on these parameters was not warranted. The primary effort in this report was directed toward the most important parameters for soil action level calculations with RESRAD: mass loading, soil-to-plant transfer factors, distribution coefficients, area of contamination, and mean annual wind speed.

Task 5 of this project, *Independent Calculations*, will use the values and distributions presented here in the calibrated version of RESRAD. Values for soil action level and dose will be presented as distributions of possible values for each individual scenario.

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APPENDIX A:
JOINT FREQUENCY TABLES

FOR FIVE YEAR AVERAGE ROCKY FLATS WIND SPEEDS

Stability Class A

Fraction of total meteorological data in stability class = 0.0133

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0	0.01504	0	0	0	0	0.01504	2.3
NNE	0	0.04511	0	0	0	0	0.04511	2.3
NE	0	0.09774	0	0	0	0	0.09774	2.3
ENE	0.00752	0.18045	0	0	0	0	0.18797	2.23799
E	0.01504	0.18045	0	0	0	0	0.19549	2.18075
ESE	0	0.18045	0	0	0	0	0.18045	2.3
SE	0	0.10526	0	0	0	0	0.10526	2.3
SSE	0	0.03759	0	0	0	0	0.03759	2.3
S	0	0.03008	0	0	0	0	0.03008	2.3
SSW	0	0.03008	0	0	0	0	0.03008	2.3
SW	0.00752	0.03008	0	0	0	0	0.0376	1.99
WSW	0	0.00752	0	0	0	0	0.00752	2.3
W	0	0.00752	0	0	0	0	0.00752	2.3
WNW	0	0.01504	0	0	0	0	0.01504	2.3
NW	0	0	0	0	0	0	0	0
NNW	0	0.00752	0	0	0	0	0.00752	2.3
Totals	0.03008	0.96993	0	0	0	0	1.00001	2.25339

Stability Class B

Fraction of total meteorological data in stability class = 0.125
5

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00478	0.01673	0.00876	0.0008	0	0	0.03107	2.68233
NNE	0.00637	0.04064	0.02789	0.00159	0	0	0.07649	2.91870
NE	0.01275	0.05976	0.0239	0	0	0	0.09641	2.54123
ENE	0.01833	0.06614	0.01594	0	0	0	0.10041	2.30279
E	0.02231	0.07809	0.02311	0.0008	0	0	0.12431	2.38476
ESE	0.02311	0.10279	0.03187	0	0	0	0.15777	2.43656
SE	0.01833	0.07012	0.04382	0.0008	0	0	0.13307	2.70568
SSE	0.01195	0.04064	0.02311	0.0008	0	0	0.0765	2.64765
S	0.01275	0.02709	0.01116	0.0008	0	0	0.0518	2.37423
SSW	0.01036	0.01514	0.00717	0	0	0	0.03267	2.20352
SW	0.00956	0.00717	0.00159	0.0008	0	0	0.01912	1.85878
WSW	0.00876	0.00717	0.00398	0	0	0	0.01991	1.97785
W	0.01195	0.00956	0.00637	0.0008	0	0	0.02868	2.17669
WNW	0.00717	0.00637	0.00239	0.0008	0	0	0.01673	2.10325
NW	0.00637	0.00478	0.00319	0.0008	0	0	0.01514	2.25961
NNW	0.00558	0.00717	0.00637	0.0008	0	0	0.01992	2.61812
Totals	0.19043	0.55936	0.24062	0.00959	0	0	1	2.48014

Stability Class C								
	Fraction of total meteorological data in stability class =						0.1734	
	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00173	0.01038	0.02653	0.00461	0	0	0.04325	3.81113
NNE	0.00404	0.0346	0.03979	0.00807	0	0	0.0865	3.46610
NE	0.00461	0.03172	0.0271	0.00346	0	0	0.06689	3.15002
ENE	0.00461	0.0271	0.01557	0.00173	0	0	0.04901	2.88136
E	0.00461	0.03806	0.0248	0.00404	0.00058	0	0.07209	3.12461
ESE	0.00519	0.04325	0.02941	0.00173	0	0	0.07958	2.95978
SE	0.00346	0.05133	0.05017	0.00634	0.00058	0.00058	0.11246	3.38537
SSE	0.00346	0.0496	0.05133	0.00807	0.00058	0.00058	0.11362	3.45966
S	0.00519	0.03806	0.03114	0.00519	0.00058	0	0.08016	3.23587
SSW	0.00461	0.02191	0.01384	0.00231	0	0	0.04267	2.95456
SW	0.00404	0.01615	0.01038	0.00288	0	0	0.03345	3.05019
WSW	0.00634	0.0075	0.01038	0.00519	0.00058	0.00058	0.03057	3.63840
W	0.00634	0.0173	0.01961	0.01038	0.00115	0.00058	0.05536	3.82581
WNW	0.00461	0.00865	0.01326	0.01038	0.00173	0.00173	0.04036	4.52749
NW	0.00288	0.01153	0.01845	0.00865	0.00058	0.00058	0.04267	4.08176
NNW	0.00288	0.0173	0.02595	0.00519	0	0	0.05132	3.56816
Totals	0.0686	0.42444	0.40771	0.08822	0.00636	0.00463	0.99996	3.40169

Stability Class D								
	Fraction of total meteorological data in stability class =						0.2752	
	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0	0.00291	0.0109	0.01999	0.00254	0.00218	0.03852	6.05986
NNE	0	0.00799	0.01054	0.01453	0.00109	0	0.03415	4.95745
NE	0	0.00654	0.00654	0.004	0.00073	0	0.01781	4.24430
ENE	0	0.00763	0.00436	0.00109	0	0	0.01308	3.26666
E	0	0.00836	0.00727	0.00581	0.00036	0	0.0218	4.19183
ESE	0	0.00654	0.00436	0.00254	0	0	0.01344	3.71547
SE	0	0.00908	0.00799	0.00509	0.00073	0	0.02289	4.13634
SSE	0	0.01235	0.0189	0.01344	0.00145	0.00036	0.0465	4.59522
S	0	0.01708	0.01853	0.02144	0.00218	0.00036	0.05959	4.75876
SSW	0	0.0149	0.00981	0.01199	0.00145	0.00036	0.03851	4.48088
SW	0	0.01308	0.01163	0.01708	0.00182	0.00036	0.04397	4.85451
WSW	0	0.01417	0.01381	0.03815	0.00836	0.00254	0.07703	5.87014
W	0	0.01635	0.01417	0.06323	0.03125	0.03379	0.15879	7.48100
WNW	0	0.01344	0.01417	0.11047	0.06214	0.0556	0.25582	7.93951
NW	0	0.01381	0.01453	0.05451	0.01817	0.00581	0.10683	6.48767
NNW	0	0.00872	0.01672	0.02398	0.00182	0	0.05124	5.20226
Totals	0	0.17295	0.18423	0.40734	0.13409	0.10136	0.99997	6.27112

Stability Class E

Fraction of total meteorological data in stability class = 0.1734

	Windspeed (m s^{-1}) 0.75	2.3	4.1	6.7	9.5	11	Fractional Direction Totals	Avg. windspeed (m s^{-1})
N	0	0	0.02537	0.00173	0	0	0.0271	4.26597
NNE	0	0	0.02134	0.00058	0	0	0.02192	4.16879
NE	0	0	0.01211	0	0	0	0.01211	4.1
ENE	0	0	0.0075	0	0	0	0.0075	4.1
E	0	0	0.01096	0	0	0	0.01096	4.1
ESE	0	0	0.00807	0	0	0	0.00807	4.1
SE	0	0	0.01615	0	0	0	0.01615	4.1
SSE	0	0	0.03691	0.00115	0	0	0.03806	4.17856
S	0	0	0.08535	0.00173	0	0	0.08708	4.15165
SSW	0	0	0.08016	0.00115	0	0	0.08131	4.13677
SW	0	0	\0.12111	0.00115	0	0	0.12226	4.12445
WSW	0	0	0.15398	0.00577	0	0	0.15975	4.19390
W	0	0	0.10438	0.00519	0	0	0.10957	4.22315
WNW	0	0	0.09343	0.00692	0	0	0.10035	4.27929
NW	0	0	0.10381	0.00288	0	0	0.10669	4.17018
NNW	0	0	0.08939	0.00173	0	0	0.09112	4.14936
Totals	0	0	0.97002	0.02998	0	0	1	4.17794

Stability Class F

Fraction of total meteorological data in stability class = 0.2381

	Windspeed (m s^{-1}) 0.75	2.3	4.1	6.7	9.5	11	Fractional Direction Totals	Avg. windspeed (m s^{-1})
N	0.00252	0.0168	0	0	0	0	0.01932	2.09782
NNE	0.00504	0.01638	0	0	0	0	0.02142	1.93529
NE	0.00546	0.02016	0	0	0	0	0.02562	1.96967
ENE	0.0084	0.01722	0	0	0	0	0.02562	1.79180
E	0.00882	0.02604	0	0	0	0	0.03486	1.90783
ESE	0.0084	0.02184	0	0	0	0	0.03024	1.86944
SE	0.00882	0.02898	0.00042	0	0	0	0.03822	1.96208
SSE	0.01176	0.04452	0	0.00042	0	0	0.0567	2.01111
S	0.0168	0.06174	0	0.00042	0	0	0.07896	1.99361
SSW	0.01554	0.06762	0.00042	0.00042	0	0	0.084	2.04425
SW	0.01596	0.08148	0.00042	0	0	0	0.09786	2.05493
WSW	0.01806	0.09618	0.00042	0.00084	0	0	0.1155	2.09618
W	0.0168	0.1134	0.00042	0.00084	0	0	0.13146	2.13578
WNW	0.01428	0.09282	0	0.00126	0	0	0.10836	2.14689
NW	0.01092	0.0714	0	0.00042	0	0	0.08274	2.11776
NNW	0.00798	0.04074	0	0.00042	0	0	0.04914	2.08589
Totals	0.17556	0.81732	0.0021	0.00504	0	0	1.00002	2.05388

Composite of all Stability classes								
	Windspeed (m s^{-1})						Direction Fractional Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00149	0.00890	0.01309	0.00670	6.99008	5.99936	0.03149	4.23623
NNE	0.00269	0.01779	0.01700	0.00569	2.99968	0	0.04349	3.53334
NE	0.00369	0.02089	0.01159	0.00170	2.00896	0	0.03809	2.93183
ENE	0.00519	0.02159	0.00720	0.00059	0	0	0.03460	2.51795
E	0.00589	0.02730	0.01110	0.00239	1.99644	0	0.04690	2.78689
ESE	0.00580	0.02979	0.01169	0.00099	0	0	0.04829	2.64085
SE	0.00500	0.02849	0.01929	0.00260	3.01468	1.00572	0.05580	3.04324
SSE	0.00489	\0.02819	0.02340	0.00549	4.99612	1.99644	0.06269	3.32161
S	0.00650	0.02980	0.02669	0.00730	7.00508	9.9072E	0.07109	3.36908
SSW	0.00579	0.02630	0.01999	0.00399	0.00039	9.9072E	0.05659	3.15415
SW	0.00580	0.02710	0.02630	0.00549	5.00864	9.9072E	0.06530	3.32627
WSW	0.00649	0.02910	0.03290	0.01259	0.00240	0.00079	0.08429	3.82823
W	0.00659	0.03579	0.02629	0.02040	0.00879	0.00939	0.10729	4.83505
WNW	0.00509	0.02829	0.02269	0.03380	.01740	0.01560	0.12290	5.90207
NW	0.00389	0.02340	0.02559	0.01720	0.00510	0.00169	0.07689	4.47467
NNW	0.00309	0.01609	0.02540	0.00799	5.00864	0	0.05310	3.80132
Totals	0.07799	0.39889	0.32029	0.13499	0.03800	0.02869	0.99888	3.87038