FINAL REPORT

Technical Project Summary

Radionuclide Soil Action Level Oversight Panel

February 2000

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

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Risk Assessment Corporation
417 Till Road, Neeses, SC 29107
Phone 803.536.4883 Fax 803.534.1995

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Contributing Authors

John E. Till, Ph.D., *Risk Assessment Corporation*
Kathleen R. Meyer, Keystone Scientific, Inc.
Jill Weber Aanenson, Scientific Consulting, Inc.
George G. Killough, Hendecagon Corporation
Arthur S. Rood, K-Spar, Inc.
Helen A. Grogan, Cascade Scientific, Inc.
Warren K. Sinclair, Warren K. Sinclair, Inc.

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EXECUTIVE SUMMARY

The primary objective of this project has been to review radionuclide soil action levels (RSALs) adopted by the U.S. Department of Energy, the U.S. Environmental Protection Agency, and the Colorado Department of Health and Environment in 1996 for cleanup at the Rocky Flats Environmental Technology Site. Another objective has been to recommend a technical method for independently deriving RSALs for the site. As a result of public concern about the soil action levels established in 1996, the Radionuclide Soil Action Level Oversight Panel, a group of community members with considerable experience in Rocky Flats issues, was formed. In 1998, DOE provided funds for the Radionuclide Soil Action Level Oversight Panel to select a contractor to conduct an assessment of the interim RSALs and to independently calculate RSALs for the RFETS. Through a competitive bidding process and evaluation, *Risk Assessment Corporation* (*RAC*) was selected by the Radionuclide Soil Action Level Oversight Panel to carry out the study.

RAC's methodology for determining RSALs applicable to the Rocky Flats site was based on several extensions of an earlier approach proposed by DOE/EPA/CDPHE that used the RESRAD computer program. The contract required that the work consider maximum annual dose limits of 15 and 85 mrem in any year over the next 1000 years. *RAC* adopted the 15 mrem y^{-1} limit for a technically based RSAL because it was more protective of the public. Although several computer codes were considered for use as the basis of *RAC*'s analysis, the RESRAD code was adopted because it was the most practical choice and was required to be used in addition to any other code that may have been selected. *RAC* designed extensions to RESRAD to include (1) considering the heterogeneity of radionuclide concentrations in soil around the site, (2) quantifying uncertainty in predictions of dose, (3) considering additional exposure scenarios, and (4) treating the possible occurrence of a large grass fire. The exposure pathways considered were inhalation, soil and food ingestion, and external irradiation. In addition, groundwater use for both irrigation and drinking water was assumed for some scenarios.

The RSAL values include estimates of the uncertainties and are designed to ensure that the permitted annual dose limit for the targeted individual is exceeded only with low probability. For each scenario, curves were presented that representing the probability of exceeding the radiation dose limit as a function of $^{239+240}Pu$ or uranium concentrations in the soil. Each probability level corresponds to a distinct concentration of $^{239+240}Pu$ or uranium in soil.

RAC applied this methodology to the Rocky Flats data using the most restrictive exposure scenarios approved by the Rocky Flats Soil Action Level Oversight Panel and assuming a 10% probability that the 15 mrem y^{-1} dose limit will be exceeded (i.e. a 90% probability that the dose limit will <u>not</u> be exceeded). Using this approach, the technically derived RSAL for $^{239+240}Pu$ in soil at Rocky Flats would be 35 pCi g^{-1}. This calculation was corroborated by an alternate method calculation that also resulted in an RSAL at the 10% level of about 37 pCi g^{-1}, suggesting 35 pCi g^{-1} as a technically based RSAL for the Rocky Flats site. The results as presented are a reasonable indication of RSAL magnitudes based on purely scientific considerations if the prescribed dose is not to be exceeded.

The calculation of uranium RSALs was done somewhat differently than those for plutonium because of significant differences in the nature and extent of contamination and the mobility of uranium in the subsurface. For each uranium scenario, consideration was given to whether groundwater was a viable pathway. A viable groundwater pathway assumed that the surficial

aquifer underlying the site would provide enough water for human consumption and irrigation. The impacts of a probabilistic fire were also evaluated but inclusion of this process in our calculations made little difference in the resulting RSALs. Assuming the groundwater pathway was viable and a 10% probability, the technically derived ^{238}U RSAL for the most restrictive scenario (child of rancher) was 10 pCi g^{-1}.

We believe the general approaches presented in this report and these results are sound and we recommend their adoption. Data limitations impose uncertainties on estimates of doses, and we have been careful to indicate these uncertainties in our analysis. The project's time and budget goals precluded a more in-depth investigation of several important areas of research that, if addressed in the future, could strengthen this analysis. We have presented these recommendations for further research and recognize that they could change these results somewhat and improve them as a basis for decision making.

Public involvement was particularly important in this study because of the impact the cleanup may have on the local communities surrounding the site. *RAC*, along with the Rocky Flats Soil Action Level Oversight Panel, were committed to ensuring that there was public involvement and interaction during the entire review process through open technical work sessions and general public meetings.

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for Rocky Flats site. However, the final decision on setting the RSALs ultimately lies in the hands of the stakeholders, U.S. Department of Energy, State and federal authorities, and the community working together to arrive at a cleanup level that provides long term protection of the public. *RAC* believes that additional research in specific areas could reduce some of the uncertainties and help to develop more well-defined methods in the approach.

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TECHNICAL SUMMARY

INTRODUCTION

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. The RFETS is located 5–6 mi (8–10 km) from the cities of Arvada, Westminster, and Broomfield, Colorado, and 16 mi (26 km) northwest of downtown Denver, Colorado (Figure 1). For most of its history, the Dow Chemical Company operated the Rocky Flats Plant as a nuclear weapons research, development, and production complex. For almost 40 years the site manufactured components for nuclear weapons, and, in the process, released contaminants to the environment. In 1989 Rocky Flats stopped weapons parts production and, in 1992, began the process of cleaning up contamination at the site. The soil on the Rocky Flats site is contaminated with plutonium and uranium from routine and accidental releases of radionuclides during operations, and from leaking barrels of contaminated oils and solvents that were stored at the 903 Area, an outdoor area directly east of the main buildings at the Rocky Flats plant.

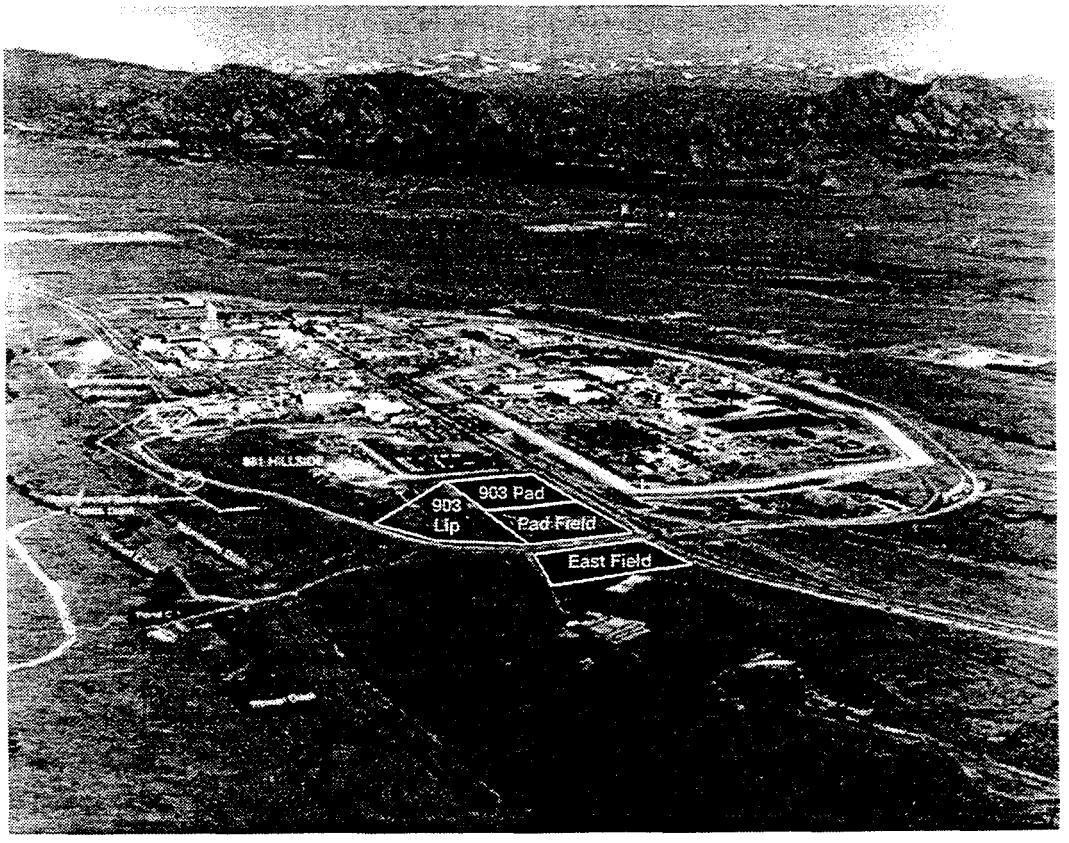

Figure 1. Aerial photograph of the Rocky Flats Plant looking northwest. The highest levels of plutonium in the soil are found in the vicinity of the 903 Pad, which is marked on the photograph.

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The focus of the current project was to develop a methodology for determining radionuclide soil action levels (RSALs) and to calculate RSALs for Rocky Flats by applying this methodology. RSALs are certain levels or concentrations of one or more radionuclides in soil above which remedial action should be considered so that people do not receive radiation doses above permitted levels. In October 1996 DOE, the U.S. Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE) adopted radionuclide soil action levels to be used in the cleanup of the Rocky Flats site (DOE/EPA/CDPHE 1996).

In response to public concern about the interim soil action levels proposed in 1996, DOE provided funding for an independent assessment and calculation of soil action levels for the Rocky Flats cleanup work.

As a result of public concern about the soil action levels established in 1996, the Radionuclide Soil Action Level Oversight Panel (RSALOP) was formed. The RSALOP is a group of community members with considerable experience in Rocky Flats issues (see Appendix E). In 1998, DOE provided funds for the RSALOP to select a contractor to conduct an assessment of the interim RSALs and to independently calculate RSALs for the RFETS. Through a competitive bidding process and evaluation, *Risk Assessment Corporation* (*RAC*) was selected by the RSALOP to carry out the study. Work began in October 1998 and was completed in March 2000.

This final report summarizes *RAC*'s work on the soil action level project. It provides an overview of the project and includes as attachments the full set of technical reports issued as a part of this project. This report outlines the background to the project and scope of work. It describes the methodology that we developed for determining RSALs, and identifies the results we obtained for each scenario. This report also describes the individual project tasks and how they contributed to the project as a whole. Finally, we provide technically derived RSALs for $^{239+240}$Pu and uranium in soil at Rocky Flats and discuss our conclusions concerning the methodology and its application to soil cleanup at RFETS.

OBJECTIVES AND DESIGN OF THE STUDY

The objective of this current study was to conduct an independent assessment of the interim RSALs adopted by DOE/EPA/CDPHE in October 1996 (DOE/EPA/CDPHE 1996) and to develop and apply a methodology for determining RSALs applicable to RFETS.

The scope of work dictated a number of design objectives for the methodology:

1. To base the soil action level on a dose limit, rather than a level of risk.
2. To consider two dose limits: 15 mrem in a year (15 mrem y^{-1}) for unrestricted use of the site, and 85 mrem in a year (85 mrem y^{-1}) for unrestricted use after failure of land use controls at the site These dose limits are those chosen for the 1996 assessment (DOE/EPA/CDPHE 1996) and are based on Draft Title 40 CFR 196. *RAC* developed technically based RSALs using the 15 mrem y^{-1} dose limit for two reasons: it is more protective of the public and our evaluation of risk associated with this dose better corresponds to the target level of risk associated with federal guidance.
3. To consider any realistic scenarios of exposure for the future, without being restricted to previously proposed scenarios.
4. To include uncertainties in the calculation to the greatest extent possible.

5. To incorporate site-specific data into the calculation where they are available.
6. To evaluate different computer codes available for calculating RSALs and select one that is the most appropriate for the situation at Rocky Flats. The RESRAD environmental transport computer code (Yu et al. 1993) was used in the previous assessment as specified by DOE Order 5400.5.
7. To use a documented and reviewed computer code, though the code could be modified to improve the quality of the calculation.
8. To evaluate all input parameters to the RESRAD computer code and suggest alternatives if values are not appropriate for the Rocky Flats site.
9. To complete the work within the time constraints given and interact with the RSALOP and the public at monthly availability sessions and formal meetings.

The dose limits for the project, 15 mrem y^{-1} and 85 mrem y^{-1}, are based on draft EPA guidance and are the dose limits used in the previous DOE/EPA/CDPHE assessment in 1996. The 15 mrem dose limit applies to a site that is open to the public and is the more important limit for this project.

The work was broken into eight project tasks that we describe in detail below. First we outline a number of other factors that influenced the *RAC* methodology. *RAC* recognized that the goal of soil action levels is to protect people who may, in the near or distant future, come into contact with a site where the soil is contaminated with radionuclides at levels above background. Therefore, setting soil action levels must consider how:

- Particular radioactive materials are transported in the environment to people (transport pathways)
- People might be exposed to the radioactive materials (exposure scenarios)
- Radiation dose to a person is assessed (radiation dosimetry)
- Radiation protection guidelines are applied (annual dose limits).

Because of these considerations, *RAC* concentrated on several processes important in the transport of radioactive materials in air and water in an area like Rocky Flats and developed exposure scenarios for the project. In designing the scenarios, *RAC* followed the principle that if the person living onsite full-time is protected, then the person living offsite will be protected. It was also important to understand the behavior of radionuclides in the soil and how soil can be disturbed or resuspended, because inhalation can be one of the important exposure pathways for those living on or near the site. The potential significance of the complex groundwater pathway was considered during the project. *RAC* did not assess the groundwater pathway in detail in the RSAL project because too little is now known. Extensive ongoing research by the Actinide Migration Evaluation (AME) team is expected to provide data not currently available. Although the groundwater pathway was ruled out in the previous analysis (DOE/EPA/CDPHE 1996), *RAC* included the pathway in our analysis for three scenarios to

Radionuclide soil action levels can be calculated for individual radionuclides in the soil. In reality, the soil contains a mix of radionuclides that can contribute to radiation dose; therefore, a mathematical method, called the sum-of-ratios, was used.

provide a likely-conservative calculation because the groundwater pathway could not clearly be ruled out as a possible exposure pathway for the future. We recognized and pointed out, however, that our assessment of the groundwater pathway was limited by the complexity of the pathway and a lack of available information.

RSALs can be calculated for an individual radionuclide, as though it were the only one present in the soil. Realistically though, the soil contains a mix of radionuclides that collectively contribute to dose. A mathematical approach, called sum-of-ratios, was used to combine the individual RSALs and assess the mixture of the contaminants in the soil. The concentration of each radionuclide is divided by its RSAL, and these ratios are added. The dose limit is exceeded if this sum-of-ratios is greater than 1. Conversely, the dose limit is not exceeded if the sum-of-ratios is less than or equal to 1

Each soil action level can be calculated in two ways: deterministically or stochastically (where uncertainties are considered). When calculated deterministically, the soil action level represents a single number, without indication of uncertainty in the value. In this case, when the ratios of radionuclide levels divided by soil action levels are summed and compared with 1, the sum-of-ratios is itself a deterministic quantity; that is, a single number, with typically no indication of uncertainty. This is the approach used in the interim soil action level calculations (DOE/EPA/CDPHE 1996).

A deterministic calculation yields a single number because the input values are single numbers with the result or output from the calculation being a single number. In contrast, a stochastic calculation takes into account the uncertainties of the input parameters, and it results in a distribution of possible results. We adopted the stochastic approach in this study.

However, the movement of each radionuclide through environmental media and into possible contact with people is an uncertain process, and mathematical models are used to simplify these processes.

In a stochastic calculation, the natural variability and lack of complete knowledge about the parameters can be treated as parameters with probability (or uncertainty) distributions. In this case, the soil action levels and sum-of-ratios that result from the model calculations, reflect the uncertainty of the input parameters. The data and input parameters are presented as probability distributions, using probability theory and Monte Carlo computational methods to propagate uncertainty to the results. Many simulations are carried out using random sampling to select values from the distributions of model parameters. This simulation yields a range of values that are used to construct uncertainty distributions for the results.

When uncertainties in soil action levels are considered, the decision about the extent of cleanup is not as straight-forward as in the deterministic case, where the sum-of-ratios is a single number that is to be compared to 1. When uncertainties are considered in the calculation, the sum-of-ratios is a distribution of values, which provides an estimate of how probable it is that the sum-of-ratios exceeds 1. If that probability is small, then a decision may be made that no action is required, even though there is some possibility that the annual dose limit could be exceeded.

Based on *RAC*'s methodology, input parameters, and exposure scenarios, we provided a technically derived RSAL for $^{239+240}Pu$ and uranium in soil at Rocky Flats. The RSAL values include estimates of the uncertainties and are designed to make it unlikely that a scenario subject would receive more than a 15 mrem y^{-1} dose. By protecting the most conservative individuals

described by the scenarios, the RSALs should protect others as well. The technically derived RSAL values selected from this methodology are those calculated to ensure that the permitted annual dose limit for the targeted individual is exceeded only with low probability.

PREVIOUS CALCULATIONS OF INTERIM SOIL ACTION LEVELS

Table 1 presents the interim soil action levels adopted in 1996 (DOE/EPA/CDPHE 1996) for individual radionuclides, assuming the presence of no other radionuclides. To represent RSALs based on a mix of radionuclides in the environment, the sum-of-ratios calculation, described earlier, must be completed. DOE/EPA/CDPHE completed an example of this type of calculation in their 1996 report, using a fixed ratio of ^{239,240}Pu and ^{241}Am. Table 2 shows the results of these calculations. *RAC*'s calculations inherently include the sum-of-ratios assumption of shared residence of contaminants in soil.

Table 1. Individual Radionuclide Soil Action Levels (in pCi g^{-1}) Adopted by DOE/EPA/CDPHE in October 1996 [a]

	Resident		Office worker	Open space user
Radionuclide	15 mrem[b]	85 mrem[b]	15 mrem[b]	15 mrem[b]
Americium-241	38	215	209	1283
Plutonium-239	252	1429	1088	9906
Plutonium-240	253	1432	1089	9919
Uranium-234	307	1738	1627	11500
Uranium 235	24	135	113	1314
Uranium-238	103	586	506	5079

[a] Taken from Table 5-1, DOE/EPA/CDPHE (1996). These RSALs were calculated using a different methodology than the RSAL values that *RAC* calculated and presents in this report. Thus, comparing the results of the two methods is misleading.

[b] Annual dose limits from EPA draft 40 CFR 196 (EPA 1996) that were used in the previous assessment (DOE/EPA/CDPHE 1996).

Table 2. Example of the DOE/EPA/CDPHE Radionuclide Soil Action Levels (in pCi g^{-1}) Based on the Sum-of-Ratios [a]

	Resident		Office Worker
Radionuclide	15 mrem[b]	85 mrem[b]	15 mrem[b]
Plutonium-239,240	115	651	562
Americium-241	21	117	101

[a] Taken from Table 5-2, DOE/EPA/CDPHE (1996), which states that "this example assumes that the ^{241}Am/^{239}Pu activity ratio equals 0.18 and that only ^{239}Pu and ^{241}Am are present." These RSALs were calculated using a different methodology than those *RAC* calculated and presents in this report. Thus, comparing the results of the two methods is misleading.

[b] Annual dose limits from EPA draft 40 CFR 196 (EPA 1996) that were used in the previous assessment (DOE/EPA/CDPHE 1996).

RAC's approach to calculating RSALs and the approach employed in the previous DOE/EPA/CDPHE (1996) assessment cannot be readily compared because the two assessments used quite different assumptions. The two sets of calculations were performed with different dose conversion factors and different soil resuspension models and data. Additionally, in the DOE/EPA/CDPHE (1996) calculation, the principal pathway contributing to a person's dose was inhalation; in the corresponding *RAC* estimate, ingestion played a much more significant role. Finally, the DOE/EPA/CDPHE calculation was deterministic, whereas *RAC*'s was stochastic; the RSALs provided by *RAC* represent the 90th percentile of a stochastic simulation. *RAC* also included the effects of a prairie grass fire in the calculation of soil action levels for each scenario, considering the probability of a fire occurring in the area (see *Task 4: Methodology for Determining Soil Action Levels* section). Evaluating similarities or differences between *RAC's* RSAL values and those reported in the DOE/EPA/CDPHE (1996) is inappropriate because of the numerous differences between the methodologies used.

PROJECT TASKS

To calculate soil action levels for the RFETS, the current project was designed to follow a careful and systematic course. The selected approach proceeded after selecting a computer code to analyze exposure pathways and sensitive parameters affecting the final results. This project was laid out in eight tasks. Five of the tasks (Tasks 1, 2, 3, 5, and 6) resulted in written reports; one task was completed as a presentation (Task 4); and two tasks (Tasks 7 and 8) were ongoing throughout the project. Table 3 summarizes the project tasks.

Each task listed in Table 3 was an important step in the process of this work. The following pages explain each task in detail. Our discussion begins with Task 8, and is followed by Task 7, because these two tasks involved interaction throughout the project.

Table 3. Project Tasks and Reports

Task	Reference	Location in current report[a]
1: Cleanup Levels at Other Sites	Weber and Till, 1999	Attachment A
2: Review Computer Models to Calculate Soil Action Levels	Killough et al. 1999	Attachment B
3: Inputs and Assumptions	Aanenson et al. 1999	Attachment B
4: Methodology for Determining Soil Action Levels	Presented at November 1998 RSALOP meeting; described in Killough et al. 2000	Attachment B
5: Independent Calculation of RSALs	Killough et al. 2000	Attachment B
6: Soil Sampling Protocol	Thorne and Rood 1999	Attachment C
7: Interaction with Actinide Migration Evaluation team	No report required[b]	Attachment D
8: Public Interaction	No report required	

[a] The layout of the final summary report was specified in the project contract; this required that the reports for Task 2, 3, and 5 be grouped into Attachment B.
[b] Attachment D contains written summaries of the quarterly AME meetings.

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TASK 8: PUBLIC INTERACTION

Public involvement was particularly relevant in this study because of the impact the cleanup levels may have on the local communities surrounding the site. With any study that involves members of the public as stakeholders, it is important to involve the public in new and creative ways in the decision-making process. *RAC*, along with the RSALOP, focused on this end throughout the project by scheduling and conducting public meetings and making written materials available. Advanced Integrated Management Services, Inc. (AIMSI) provided management support to the RSALOP for the project. The Citizens' Advisory Board managed the DOE grant for the work, made their web site available for the dissemination of project information, and provided technical assistance throughout the project.

It was important to keep the public and local stakeholders informed about the scientific review of the interim RSALs and calculations undertaken by *RAC*. The RSALOP, as well as *RAC*, were committed to ensuring that there was public involvement and interaction during the entire review process. Monthly meetings were scheduled to update the RSALOP on the progress of the project. The monthly RSALOP meetings were important forums for interaction with the RSALOP, the agencies, and the public. During these meetings, *RAC* received guidance from RSALOP members regarding the direction of the project and input parameter values. Ideas and insights coming from discussions at these meetings would have been lost had this interaction not existed.

The RSALOP and *RAC* agreed to hold informal public technical sessions immediately before the regular monthly meetings to make sure all questions could be addressed and all issues discussed fully. These technical work sessions gave anyone interested in attending a chance to ask more specific questions and to discuss the technical details of the work. These sessions served as round-table discussions and much was accomplished by explaining our methodology, clarifying issues, and presenting examples of our work to the attendees.

Three general public meetings took place during the course of the project. The public meetings were geared toward a general audience to update them on the work being done and to respond to their questions and comments.

A special workshop on risks from exposure to radioactive materials was held on February 11, 1999, in response to panel concerns about radiation risk estimates and how they are derived. This workshop, led by Mr. Charles Meinhold, President of the National Council on Radiation Protection and Measurements, was open to the public and helped inform the RSALOP about current practices in risk protection and management.

Five peer reviewers from around the country, contracted by the RSALOP and paid from community funds, reviewed and provided written comments on each of our five draft technical reports. We also received review comments from RSALOP members, DOE personnel, and others attending the monthly meetings. This review process helped identify areas of concern not already considered and allowed us to address many of the concerns within the context of our work. We responded to all comments in writing, and these responses were reviewed and accepted by the RSALOP. The final reports reflected changes made in response to all review comments. The process of public interaction and review took place throughout the entire project and provided a valuable means for identifying issues that were critical for the public.

TASK 7: ACTINIDE MIGRATION EVALUATION AT ROCKY FLATS

The Actinide Migration Studies group was established by DOE in 1996 to investigate and model $^{239,240}Pu$, ^{241}Am, and uranium transport in the site environment. (The project was later renamed the Actinide Migration Evaluation [AME]). Periodic technical and public meetings have enabled the scientists involved in the study of actinide movement in the Rocky Flats environment to report on the progress of their work. Their input into our project was valuable , and Attachment D contains summaries of meetings held during the course of the project.

Quarterly AME working meetings were held throughout the duration of the RSAL project, and *RAC* attended the meetings to gather information that might be helpful for our studies. Research on actinide migration processes at Rocky Flats is relevant to the current soil project because the studies may help characterize the chemical and physical form of plutonium at the Rocky Flats site and help define the potentially significant pathways of exposure. To date the research into the movement of actinides in the Rocky Flats environment has not provided any new models of groundwater movement, but it did provide *RAC* some information to make our calculations more site-specific. For example, surface water discharge and actinide activity data from site monitoring programs during the 1990s were compiled to compute actinide loads on a storm-specific and annual basis. Other studies report $^{239,240}Pu$ and ^{241}Am activity in surface soil sampled in the Walnut and Woman Creek watersheds. These types of data can be used to calibrate the models that estimate soil erosion and associated actinide transport. Other work is underway to characterize plutonium in samples from the 903 Area using state-of-the-art analytical techniques. This work has so far concluded what many have assumed all along—that plutonium in the soil at Rocky Flats is insoluble PuO_2. While results from some of the AME studies indicated that this insoluble form of plutonium might not enter groundwater, *RAC* used a conservative approach to address the question of whether or not the groundwater exposure pathway could be ruled out of the current analysis. We understood the importance of groundwater and surface water pathways in the long term and we included the groundwater pathway in one of our scenarios. We did recognize, however, that our assessment of the groundwater pathway was limited by the complexity of the pathway and, therefore, we used broad ranges of values for the soil-water equilibrium distribution coefficients.

Findings from the Actinide Migration Evaluation studies were incorporated into our calculations and results. Studies verified that (1) plutonium is in an insoluble form in the soil in the 903 Area, an outdoor storage area that is heavily contaminated and that (2) actinides move through the Rocky Flats environment quite slowly toward the aquifer, if they move at all.

The AME work provided two key pieces of information to the current study. First, the AME research provided a qualitative picture of movements of actinides through the Rocky Flats environment; that they move quite slowly toward the shallow aquifer underlying the site if they move at all. Continuing research will hopefully solidify understanding of these processes and perhaps produce a more complete model for actinide movement in the Rocky Flats environment. Second, some of the AME's research was adapted for use in our independent calculations of soil action levels. We used the work of Honeyman and Santschi (1997) to provide upper bounds on our distributions of the soil-water equilibrium distribution coefficient. This quantity, in a simple

dissolved-phase transport model, dictates the rate of actinide movement. It is possible that future AME work might have an impact on *RAC*'s results.

TASK 1: CLEANUP LEVELS AT OTHER SITES

This task was designed to provide the RSALOP with a clear and unbiased comparison between soil action levels previously developed at other sites and those adopted for Rocky Flats in October 1996 (Weber and Till, 1999). Because of the varied analytical methodologies, cleanup criteria, exposure pathways, and input parameters for calculating dose and contamination levels in soil, this task was challenging, but it provided an important comparison and perspective on the magnitude of the interim soil concentration criteria at Rocky Flats. The Task 1 report, which is included as Attachment A in the current report, describes information for each site in terms of the dose, scenario, and potential exposure pathways used to calculate the cited soil action level or cleanup target. The study evaluated $^{239,240}Pu$ soil action levels that were reported to be protective of human health based on reasonable land use scenarios and predetermined dose criteria. Soil concentrations of ^{241}Am were also provided when available for these sites or facilities.

To compare cleanup levels or soil action levels, at other sites with those at Rocky Flats, RAC normalized the levels to the radiation dose that would result from being exposed to each soil concentration.

The sites methods included in the Task 1 analysis were

- Rocky Flats Environmental Technology Site, Colorado
- Hanford, Washington
- Nevada Test Site, Nevada
- U.S. Nuclear Regulatory Commission (NRC) codes for remediation
- Johnston Atoll, Marshall Islands
- Enewetak Atoll, Marshall Islands
- Maralinga, Australia
- Semipalatinsk Nuclear Range, Kazakhstan
- Thule, Greenland
- Palomares, Spain.

Dose is a general term denoting the quantity of radiation or energy that is absorbed by the body. Effective dose provides a measure of the dose to the whole body, taking into account the dose absorbed by each of the target organs and the sensitivity of those organs to radiation.

To provide an equitable comparison among the sites, the reported soil action levels or cleanup criteria were normalized to the targeted effective radiation dose after cleanup was completed. This procedure resulted in a ratio for each site, calculated by dividing the reported soil action levels or cleanup criteria by the radiation dose. In the report, we refer to this ratio as the soil action level to dose ratio. A lower soil action level to dose ratio indicates a lower proposed or calculated cleanup level. This ratio enabled us to identify the factors or parameters that affected the outcome of the calculation to the greatest extent and that accounted for the differences among soil action levels at the different facilities. Table 4 summarizes the results of our analysis, and appears as a summary table in the report (Attachment A, Weber and Till 1999).

Table 4. Summary of Comparisons between Rocky Flats Environmental Technology Site Calculations and Those for Other Facilities [a]

Location	Parameter change	Ratio of the soil action limit to dose ([pCi g^{-1}] $mrem^{-1}$)	Ratio of the dose to soil action limit (mrem [pCi g^{-1}] $^{-1}$)
Rocky Flats residential	Original calculation	17	0.06
Hanford residential	Original calculation	2.3	0.44
	Remove meat, milk, fish, drinking water pathways; change to RFETS dose conversion factor[b] and mass loading	34	0.03
Rocky Flats office worker	Original calculation	73	0.01
Hanford industrial worker	Original calculation	16.3	0.06
	Change dose conversion factor[b] and mass loading	159	0.006
Rocky Flats residential	Original calculation	17	0.06
	Change to Nevada Test Site dose conversion factor[b]	2.8	0.36
Nevada Test Site residential	Original calculation	4.1	0.24
Rocky Flats office worker	Original calculation	73	0.01
	Change dose conversion factor	16	0.06
Nevada Test Site industrial worker	Original calculation	41	0.02
Rocky Flats	Original calculation	17	0.06
Johnston Atoll	Original calculation	0.85	1.2
	Change to RFETS mass loading, enrichment factor and calculate air concentration using RFETS dose conversion factor and breathing rate	17.8	0.056
Rocky Flats	Original calculation	17	0.06
Maralinga	Original calculation	0.56	1.8
	Change to RFETS mass loading, breathing rate, dose conversion factor	17.8	0.056
Rocky Flats	Original calculation	17	0.06
	Change to Palomares breathing rate	14.1	0.07
Palomares	Original calculation	12.3	0.08

[a] From Weber and Till (1999); see Attachment A.
[b] Dose conversion factor is the committed effective dose per unit intake of radioactivity through exposure pathways like inhalation or ingestion; the impact of the dose conversion factor on the calculations is explained more fully in the *Task 5: Independent Calculation* section.

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Our evaluation showed that the interim soil action levels at the RFETS (DOE/EPA/CDPHE 1996) are higher than action or cleanup levels at other facilities, even when normalized to dose. Our comparison was done using the RESRAD Model Version 5.61, which was used to set the interim RSALs in October 1996. We reviewed the soil action level to dose ratios for the other sites in terms of the calculations, models, and parameters used to calculate soil concentrations and/or dose. The outcome of the RESRAD calculation was strongly controlled by a few parameters, and, almost without exception, it was these parameters that affected the differences in the soil action levels for a unit dose between sites. If the same or similar assumptions were made for each site, similar ratios resulted. The parameters that affected the determination of soil action levels or clean-up criteria to the greatest extent were the

- **Dose conversion factor (solubility class of plutonium):** The dose conversion factor represents the committed effective dose per unit intake of radioactivity through exposure pathways like inhalation or ingestion. For plutonium, the dose conversion factor depends, to a considerable extent, on the assumed solubility of the plutonium. For example, soluble plutonium has a dose conversion factor for inhalation that is about 1.4 times greater than for insoluble plutonium; more importantly, for ingestion, the dose conversion factor for soluble plutonium is over 65 times higher than for insoluble plutonium.[1] These differences mean that the form of the plutonium in the soil assumed for each site (i.e., soluble or insoluble) greatly impacted the level of cleanup that was done or required. The difference in the chemical form of the plutonium in the soil accounted for the difference in the cleanup standards at several of the sites with lower cleanup standards than at the RFETS. For example, the plutonium in the soil at the Hanford site was assumed to be soluble while the plutonium at the RFETS site was assumed to be in an insoluble form. When we did a calculation based on the assumption of soluble plutonium for the RFETS, the ingestion pathway became a more dominant contributor to the dose, and the dose per unit intake was considerably greater.

- **Mass loading (resuspension)**: The mass loading parameter, a measure of the resuspension of material transferred from the soil surface to the atmosphere, can vary over orders of magnitude depending on the assumed environmental conditions. Mass loading and similar resuspension parameters have been extensively measured at Rocky Flats under a variety of conditions.
- **Breathing rate**: The breathing rate of the exposed individuals has a less pronounced effect on the cleanup or soil action levels than the previous two parameters because the range of possible values is limited to within reasonable physiological boundaries.

A more complete evaluation of the primary model input parameters and assumptions is described and summarized in the full report for Task 3, *Inputs and Assumptions* (Attachment A; see also Aanenson et al. 1999).

[1] The dose conversion factors referred to here are the ICRP 30 dose conversion factors that were used in the different analyses reviewed for Task 1. The newer ICRP 67 and 71 dose conversion factors are used by *RAC* in the current analysis.

TASK 2: REVIEW AND SELECTION OF COMPUTER MODELS

Task 2 focused on reviewing the RESRAD computer model used to calculate the interim soil action levels adopted for Rocky Flats. It also reviewed four other computer-based models that could potentially be used for making calculations of soil action levels for the RFETS (Killough et al 1999). The objective was to select the most suitable one for our analysis.

The models reviewed were RESRAD, MEPAS, GENII, MMSOILS, and DandD. DOE calculated the 1996 Rocky Flats soil action levels with the RESRAD program (Version 5.61), and part of the scope of this project was to review their calculations for choice of the parameter values used in RESRAD. *RAC* selected programs that were generally comparable to RESRAD and that are widely used. All five programs examined were developed under sponsorship of one or more federal agencies. The results of this discussion and comparison of models are contained in Killough et al (1999), Attachment B to this report.

RAC selected the programs using the following criteria:

1. Correctness of the mathematical models. How well does the model account for exposure pathways and site features, and how consistent is the program with site-specific data?
2. Validation of the programs. Has the program been checked or confirmed with data that are well documented?
3. Source code. How available is the entire computer code to *RAC,* and has the program been documented?
4. Platform (i.e., computer and operating system) and programming language.
5. Flexibility of operating features. Is it possible to bypass the graphic user interface to directly specify input and output files from the operating system level?

A further consideration in selecting computer programs for the study was our desire to use state-of-the-art methods for carrying out our work, particularly by incorporating uncertainty estimates into the process of calculating RSALs. The term uncertainty usually implies lack of full or precise knowledge about the value of a model parameter or the accuracy of a model prediction. *RAC* represented these uncertainties as probability distributions. Because inputs to the selected code were in the form of probability distributions, *RAC* carefully considered the suitability of the various computer programs for providing a distribution of results for dose, or soil action levels.

Five environmental assessment computer codes (RESRAD, MEPAS, GENII, MMSOILS, and DandD) were evaluated for their applicability to calculating radionuclide soil action levels for the rocky Flats site. We concluded that either RESRAD or GENII could be adapted for the purposes of the project. We selected RESRAD.

All five of the programs selected for evaluation could be installed and executed under some version of the Microsoft Windows operating system and, as a result, all of the programs were accessible. The following paragraphs summarize our evaluation of each of these computer programs.

RESRAD (RESidual RADioactivity) was developed by DOE and Argonne National Laboratory to evaluate the cleanup and remediation of radionuclide-contaminated soils at DOE

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facilities. *RAC* used the most recent version of RESRAD (Version 5.82), which differs in some ways from older versions that are still in use. In general, the newer version is a windows-based application of earlier versions of RESRAD. The primary technical difference in the newer version, however, is how the program treats the resuspension of soil. *RAC* bypassed this portion of the code and developed resuspension factors based on site-specific data from Rocky Flats (Aanenson et al. 1999; Killough et al. 2000).

The Multimedia Environmental Pollutant Assessment System (MEPAS), which was developed at Battelle's Pacific Northwest Laboratories (PNL) and commercially marketed, was applicable to radioactive and nonradioactive pollutants in many environmental media. Because Battelle Memorial Institute declined our request for permission to examine portions of the MEPAS source code, we were not able to consider the MEPAS program for application to the Rocky Flats site. GENII, also developed at PNL, provided internal and external dose estimates for exposure through all pathways that were ordinarily considered in environmental radiological assessments. GENII had been under development for more than a decade and was unlikely to be modified further by its developers. Two resuspension models are available in GENII, including a mass loading approach that is similar to the one in RESRAD. GENII also has available a scenario of an offsite subject who has been exposed to radioactivity that has been released from the site. The RESRAD code in its traditional format cannot address such an offsite scenario. GENII also considers an onsite groundwater pathway similar to RESRAD's implementation.

MMSOILS, developed for the EPA, was a large multimedia environmental transport program that was designed for screening assessments of chemical contamination. Although it did not treat radioactivity and decay chains, it was included in our review because with some modification, it could consider radionuclides in soils. *RAC* ruled out its use in developing soil action levels for the Rocky Flats site, given the time constraints of this project.

The *Decontamination and Decommissioning* (DandD) computer program was designed by the NRC as a screening-level analysis program to provide a simplified estimate of the dose to an average member of a screening group of people. We decided against DandD because it was still in its first version and had not been used extensively and did not have published documentation. Moreover, the source code had not been released at the time our project began.

Based on our evaluation of the available computer codes, *RAC* concluded that either RESRAD or GENII could be adapted for the purposes of the project. *RAC* used the most recent version of RESRAD (Version 5.82) for this project. Attachment B contains the full Task 2 report for further details on the models.

TASK 3: INPUTS AND ASSUMPTIONS TO THE MODEL

Following the evaluation of computer codes, *RAC* identified and developed probability distributions for the input parameters to the soil action level calculation that had the most significant impact on dose. The selection of these values and distributions is the subject of the report for Task 3, *Inputs and Assumptions* (Aanenson et al. 1999, see also Attachment B). We also developed exposure scenarios, that is, hypothetical individuals who might be exposed to radioactive contamination at Rocky Flats in the future. These scenarios specify the individuals for whom doses and soil action levels were calculated.

To calculate RSALs, *RAC* used the most recent version of RESRAD (Version 5.82)) to calculate the soil action levels for this project. To run the code for this project, numerous input

values and assumptions needed to be selected to determine the soil action levels for cleanup at the RFETS so that the permitted annual dose of 15 mrem (in some cases 85 mrem) would not be exceeded. We performed 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 the most important to the final outcome, *RAC* developed site-specific values if data were available or created uncertainty distributions of values from published literature sources if site-specific data were not available. The probability distributions described the uncertainty in the values that arose from natural variability or from incomplete knowledge about a particular parameter. Attachment B of this report includes our assessment of the inputs and assumptions (Aanenson et al. 1999).

The sensitivity analysis was a single-parameter analysis, where a range of values for one parameter at a time was evaluated. Of over 50 parameters assessed for their influence on the final result, four parameters were found to impact the final result to the greatest extent. These parameters were:

- soil-water equilibrium distribution coefficient
- area of contamination
- mass loading factor
- mean annual wind speed.

Six parameters were found to affect the outcome of the calculation only slightly: (1) cover depth (depth of soil that must be removed to reveal the contaminated soil), (2) fraction of the total outside air contamination that is available indoors (indoor dust filtration), (3) soil-to-plant transfer factors, (4) depth of soil mixing layer (depth of uniform contamination), (5) fraction of irrigation water contaminated by groundwater, and (6) thickness of contaminated zone (non-uniformly distributed). The results showed little sensitivity to more than 40 other parameters required to run the RESRAD code, and therefore additional effort was not given to changing or revising the values from the ones used in the previous RSAL assessment., *RAC* made minor changes for some parameters, either in the value previously used or in the method of calculating the parameter value, to ensure a consistent approach.

Parameter Evaluations

Most of our efforts focused on providing parameter values or uncertainty distributions for the four most important parameters, based on site-specific data or on literature values (Aanenson et al. 1999, Attachment B). Table 5 summarizes the differences in parameter values or approach between the previous DOE/EPA/CDPHE assessment (DOE/EPA/CDPHE 1996) and our approach (Aanenson et al. 1999; Killough et al. 2000).

The *distribution coefficient* is important in the 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 *RAC* rancher scenario. The distribution coefficient can extend over a very wide range even for a single type of soil; therefore, it was important to incorporate as much data as possible in our assessment. We created a wide distribution of values for distribution coefficients of uranium, plutonium, and americium, based on an extensive review of the published literature (Honeyman and Santschi 1997; Sheppard and

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Thibault 1990; Dames and Moore 1984; Till and Meyer 1983). In our assessment, the distribution for each radionuclide was defined by the geometric standard deviation, which gives an estimate of the uncertainty there is about the midpoint (geometric mean or median).

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 can vary by factors of more than 100. This large heterogeneity contradicts the uniformity that the RESRAD soil model assumes. 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 contours approximate the actual contamination in soil and were used in RESRAD to help calculate soil action levels.

Table 5. Values for the Four Most Sensitive Parameters for the Independent Calculation and Comparison with Those from the Previous Assessment[a]

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	Will use 5-year annual average STAR data set collected at Rocky Flats meteorological station

[a] From Aanenson et al. (1999); see also Attachment A.
[b] GSD = geometric standard deviation, which is a measure of the extent of the distribution

The *mass loading* parameter is 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 documented. 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 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 soil contamination data to estimate a relationship between concentrations in air and soil that was used to estimate resuspension. This process bypasses the calculation in RESRAD and defines resuspension based on actual air monitoring data from the site.

The *mean annual wind speed*, not required in the previous version of RESRAD, is important in estimating resuspension in the current RESRAD Version 5.82. However, *RAC*'s method bypasses the RESRAD calculation of resuspension. Because we estimated resuspension based on site-specific air monitoring data, it also was important to use site-specific meteorological data. *RAC* used 5-year average frequency information from the onsite Rocky Flats meteorological

station for wind speed, wind direction, and atmospheric stability class. The effect of high wind events on moving contamination from the 903 Area before it was covered with an asphalt pad was evaluated in the Historical Public Exposure Studies on Rocky Flats (Weber et al. 1999). Because high wind results 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, we did not evaluate high wind events separately in this project.

Initial concentrations of radionuclides are also important parameters for RESRAD if dose is to be calculated (computed RSALs do not depend on initial soil concentrations of the corresponding radionuclides; any non-zero value given for a radionuclide will trigger the computation of the RSAL). 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 at the site to determine actual soil concentrations, initialized to the year that the soil action level calculations begin. This technique accounted for the appropriate ratios of radionuclides to the initial calculation of action levels. Because soil concentrations for uranium at Rocky Flats are primarily located in hot spots, we calculated separate soil action levels for uranium based on the concentration of uranium in hot spots, as determined from the available literature.

The amounts of the radionuclides considered in the calculation and present in soil were determined relative to $^{239,240}Pu$.

Radionuclide	*Relative Concentration*
$^{239,240}Pu$	*1*
^{241}Am	*0.111*
^{237}Np	*0.000000786*
^{238}Pu	*0.0132*
^{242}Pu	*0.00000762*

Plutonium Solubility and Dose Conversion Factors

An important aspect of the independent calculation involved determining solubility of plutonium in the Rocky Flats environment and determining dose conversion factors for use in our calculations. Ongoing studies of actinide migration at the site have helped to characterize the chemical and physical form of plutonium at the Rocky Flats site (see Attachment D). The plutonium that is found in Rocky Flats soil is thought to be generally highly insoluble and strongly attached to soil particles. Plutonium mobility is another area under investigation by the AME researchers that may play an important role at the site. These solubility studies guided the selection of dose conversion factors for plutonium and other radionuclides. Table 6 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.

Insoluble forms of plutonium are classified as slow clearance materials. *RAC* researched the most updated values available for dose conversion factors from ICRP (1999) and used them in our calculations. These newer values account for reduced uptake of plutonium from the lung based on a new respiratory tract model. The newer model accounts for changes in the relative amount of material entering the gastrointestinal-tract from the respiratory tract and also addresses the dose to specific cell populations that are at depth in the airways rather than the smeared dose used in the earlier respiratory model. In addition, dose conversion factors do show some limited age dependence. For very young infants (0-3 months), the ingestion pathway is more important,

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with a dose conversion factor for ingestion about 16 times higher than in adults. All other ages have ingestion dose coefficients less than a factor of 2 higher than the adult values.

Table 6. Dose Conversion Factors (DCF) 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 (ICRP 1978) values have been used in RESRAD Versions 5.61 and 5.82. The symbols, W (week) and Y (year) indicate the relative time required for the material to be cleared from the respiratory system.

[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])

Scenarios

RAC evaluated seven scenarios for the project. Three scenarios were developed for the original RSAL calculation (DOE/EPA/CDPHE 1996) and four scenarios were developed by *RAC* after numerous discussions with the RSALOP at the monthly soil action level meetings. Table 7 summarizes the parameter values for these scenarios. In designing the scenarios, we carefully considered offsite exposures so that if the person living onsite full-time is protected, then the person living offsite will be protected.

RAC evaluated seven scenarios designed to ensure that if the person living onsite full-time is protected then a person living offsite also will be protected. We developed four scenarios with input from the Radionuclide Soil Action Level Oversight Panel and included three scenarios from the previous DOE/EPA/CDPHE 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 focused our greatest effort on establishing values for breathing rate and soil ingestion, as these are parameters in which the RSALOP expressed primary interest. We based parameter values for breathing rate and soil ingestion on published breathing rate studies. We 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, we created distributions of scenario breathing rates. We then selected the 95th percentile value from that distribution for the annual breathing volume. We used a similar process 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 important to consider carefully the implications of translating these short-term soil ingestion rates to an annual soil ingestion rate. For these reasons, we selected the 50th percentile, or median, of the distribution as the daily soil ingestion rate for our scenarios.

Table 7. Key Scenario Parameter Values for DOE and *RAC* Scenarios[a]

Parameter	DOE/EPA/CDPHE scenarios			*RAC* scenarios			
				Nonrestrictive			
	Resident	Open space	Office worker	Resident rancher	Child of rancher (10 y)	Infant of rancher (2 y)	Site industrial worker
	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
Time on site (h y^{-1})	8400	125	2000	8760	8760	8760	2100
Time indoors onsite (%)	100	100	100	60	750	90	40
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[b]	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 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

[a] From Aanenson et al. (1999); see also Attachment B.
[b] This pathway was not applicable to this scenario.

For the remaining parameters, we used the scientific literature to select appropriate values, which in some cases differ from the RESRAD default values or the DOE/EPA/CDPHE scenarios

(DOE/EPA/CDPHE 1996). All scenario-related parameters are treated deterministically in this analysis.

TASK 4: METHODOLOGY FOR DETERMINING SOIL ACTION LEVELS

Designing a methodology for calculating soil action levels based on our exposure scenarios was the focus of Task 4. Our approach was presented to the RSALOP orally in November 1998 and documented in the Task 5 report (Killough et al. 2000; see also Attachment B). This methodology included the uncertainties in the input parameters and the resulting soil action levels. Our calculations for the RSALs were required to meet the EPA draft regulation (EPA 1996) that was chosen for the 1996 assessment (DOE/EPA/CDPHE 1996). This regulation stated that a remediation standard of 15 mrem y^{-1} should be used at sites with radioactive material in all environmental media. The radiation dose to be received by an unrestricted release scenario will not exceed 85 mrem y^{-1}. *RAC* developed technically based RSALs using the 15 mrem dose limit because (1) it is more protective of the public and (2) our evaluation of risk associated with this dose better corresponds to the target level of risk associated with federal guidance.

For a single radionuclide, scenario, and dose limit, the soil action level is that concentration of the radionuclide in the soil that would lead to a maximum predicted annual dose equal to the annual dose limit. When considering multiple radionuclides, each radionuclide's soil concentration is divided by its RSAL, and the ratios are added to give a sum-of-ratios. If the sum-of-ratios exceeds 1 for one or more of the exposure scenarios, then some remedial action or special attention is indicated. If the sum-of-ratios is less than or equal to 1, no annual dose limit would be exceeded, and by that criterion the radionuclide levels meet the RSAL standard. This calculation may be applied to observed radionuclide soil concentrations, or it may be used with hypothesized concentrations that represent remediation goals. The sum-of-ratios (SR) is indicated by the following equation:

$$SR = \sum_{i=1}^{n} \frac{C_i}{RSAL_i}$$

where
C_i = the radionuclide soil concentration for radionuclide i (pCi g^{-1})
$RSAL_i$ = soil action level for radionuclide i (pCi g^{-1})

If only one radionuclide is present, the sum-of-ratios reduces to a single ratio, but the interpretation is the same. The sum-of-ratios calculation for uranium was kept separate from that of plutonium. It was not possible to combine the two for a generic site because at Rocky Flats uranium contamination is localized and is not as widespread as plutonium.

The conceptual site model used to calculate plutonium RSALs was based on a heterogeneous distribution of plutonium soil and air concentrations across the RFETS. This conceptual model differs significantly from that used in the 1996 DOE/CDPHE/EPA calculations, which assumed soil and air contamination was homogeneous across the site being modeled.

We used the air dispersion model to incorporate soil and air concentration heterogeneity into the calculation. RESRAD was used only to calculate intakes and doses. Incorporating soil and air concentration heterogeneity into the conceptual model complicates both the calculation and

interpretation of RSALs because the RSAL depends not only on the receptor scenario parameters, but also on the location of the receptor relative to sources of contamination. Because our objective was to provide a conservative RSAL that could be applied independent of location across the RFETS, we located each receptor at the point of the maximum air-to-soil concentration ratio. The location where this occurs is at the east edge of the site near Indiana Street. Air concentrations at this location are proportionally higher than the soil concentration because the air concentrations reflect the cumulative flux from all upwind contaminated areas.

Our methodology incorporated environmental dose models that estimate dose from specified concentrations of radionuclides in environmental media. The exposure pathways considered were inhalation, soil and food ingestion, and external irradiation. In addition, groundwater use for both irrigation and drinking water was assumed for some scenarios. We also considered the effect of a prairie fire, which would remove the vegetative cover and result in increased resuspension of soil for a period of time, because such a fire, although not common, is possible. For each scenario, we incorporated the probability of a fire occurring in the area using fire statistics for the Twentieth Century in the Arapaho and Roosevelt National Forests and the Pawnee National Grasslands. For the plutonium assessment, the probability of a fire occurring on the rancher's land at the RFETS was estimated to be about 1×10^{-3}.

We calculated RSALs for uranium differently than those for plutonium because the nature and extent of contamination differs between the nuclides. Our treatment of plutonium considered a 10-km^2 contaminated area. Using spatially variable soil concentrations and measured air concentrations of plutonium around the site, we calibrated a suspension model so that the suspension rates of plutonium-contaminated soil would yield concentrations currently measured at the air samplers. This procedure was not extended to uranium because (a) uranium-specific measurements were not available at the samplers and (b) uranium contamination is not as widespread as plutonium and, therefore, would not be expected to respond in the same manner. Our investigation indicated that uranium contamination was mainly limited to past disposal areas and burn pits. Furthermore, Litaor (1995) notes fundamental differences in solubility characteristics of plutonium and uranium that, in turn, affect their mode of dispersion in the environment.

Furthermore, the prairie fire was not considered for the uranium analysis because the smallest fire area considered in the fire statistics data set was 4.05×10^5 m^2, or 100 acres. Using the area encompassed by uranium contamination (100 m^2), yields a probability of a fire that is 5 orders of magnitude lower than that for the plutonium case. Additionally, only the inhalation pathway was affected by the fire, and inhalation doses made up a small fraction of the total uranium dose. Nevertheless, we ran a trial fire case to verify that, even if there were a fire, the doses from uranium would not be significantly higher. For this trial, we conservatively assumed that any fire occurring on the site encompassed a uranium-contaminated area. Results of this trial showed incorporating the fire made little difference in the calculated dose and RSAL for uranium.

TASK 5: INDEPENDENT CALCULATION OF THE SOIL ACTION LEVELS

RAC presented the results of its independent assessment and calculation of RSALs at Rocky Flats in the Task 5 report (Killough et al. 2000; Attachment B). The Task 5 report contained details of our technical approach for determining isotopic ratios, estimating concentration of plutonium in air, calculating an alternative groundwater dose from measurements in the literature,

providing perspective on risk, and describing other computational details of the RSAL calculations. For the calculations, we used the RESRAD Version 5.82, an updated version of the RESRAD program used for the earlier calculations. We developed the methodology for selecting RSALs for the Rocky Flats site and presented the results as probability distributions of possible sum-of-ratio values for each of seven exposure scenarios. The scenarios were selected with consensus by the panel to represent a variety of exposure conditions, some of which were more conservative than others. Each scenario was based on an annual dose limit to the receptor resulting from exposure to Rocky Flats radionuclides.

For each scenario, we presented curves representing the probability of exceeding the radiation dose limit as a function of $^{239+240}Pu$ or uranium concentrations in the soil. Figure 2 shows an example of the calculational output. For example, an RSAL at the 5% to 10% probability level means there is a 95% to 90% probability that the dose limit will *not* be exceeded.

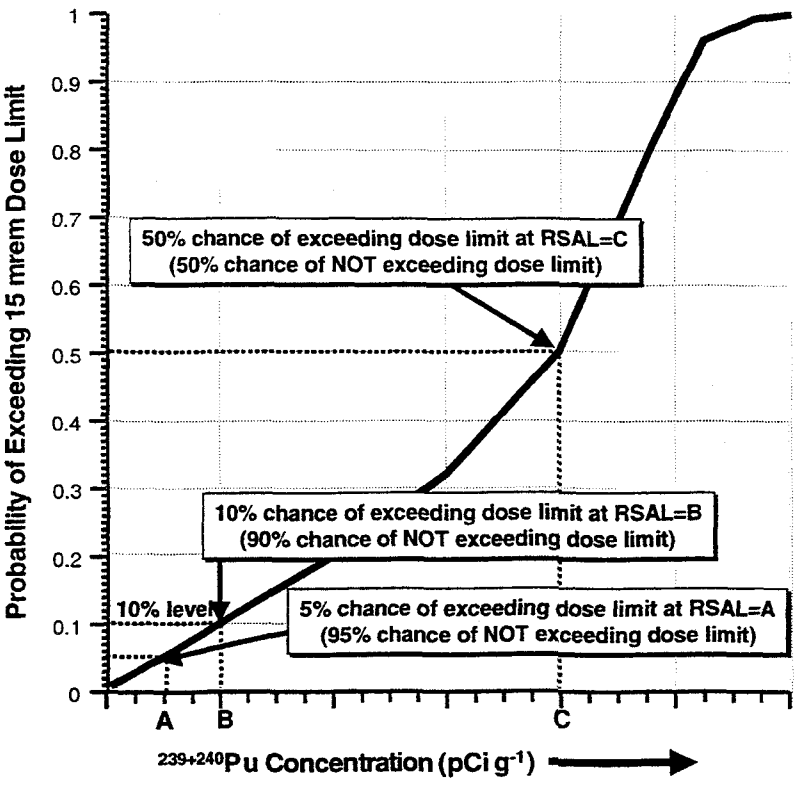

Figure 2. Sample of the results of our calculations. Each probability level corresponds to a distinct concentration of $^{239+240}Pu$ in soil. The probability value represents the probability of exceeding the dose limit. For example, at soil concentration A (measured in picocuries per gram), there is a 5% chance that some person identified by the scenario will exceed the annual dose limit. Alternately, there is a 95% chance that the dose limit for the given soil concentration will not be exceeded in any year. When we speak of probability levels throughout the report, we speak in terms of the probability of exceeding the annual dose limit.

A similar probability curve was developed for each scenario and exposure condition (Killough et al. 2000, Attachment B). RSALs are presented for plutonium isotopes for seven scenarios: the three DOE scenarios and the four *RAC* scenarios. RSALs are presented for uranium isotopes (^{234}U, ^{235}U, and ^{238}U) for three scenarios: the DOE resident (DOE-1), the *RAC* rancher (*RAC*-1), and the *RAC* child of the rancher (*RAC*-2). For the plutonium RSAL calculations, each scenario incorporated the impact of a prairie fire, considering both the probability of it occurring and the impact that revegetation might have on the soil conditions after a fire. In the Task 5 report, we also explained the scenarios, important pathways, and radionuclides contributing to dose.

For the DOE scenarios, we calculated RSALs stochastically using our methodology. It is somewhat misleading to compare the results of our calculations with the results of the DOE/EPA/CDPHE calculations (DOE/EPA/CDPHE 1996) because of differences in the methods and parameters used. For the three DOE/EPA/CDPHE scenarios, these differences are attributed to several factors. First, there was a difference in the dose conversion factors because we used the more recent ICRP Publication 72 dose conversion factors, which are higher for ingestion and lower for inhalation than the older dose conversion factors used as defaults in RESRAD (ICRP 1996). Second, the resuspension model used in our calculation results in a lower concentration of plutonium in air for a given soil concentration than the original DOE/EPA/CDPHE calculation. Consequently, the relative importance of the inhalation pathway diminishes in our calculation.

The following section highlights some of our key findings.

Plutonium: Selected Probability Curves and RSAL Values

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for the Rocky Flats site. In the following section we provide RSALs supported by the scientific data, as specified in the scope of work, However, the final decision, which must consider other factors, ultimately lies in the hands of the stakeholders, DOE, and the regulators. In addition to the important calculational aspects of the RSAL, several other criteria influence the decision-making process of selecting an RSAL for the site. Each element of the decision must be carefully considered and its importance weighed accordingly. Our approach has been to develop scientifically defensible soil action levels that both protect the public from receiving an exposure in excess of the dose limit and are reasonable to adopt, given certain social and political implications. The values we presented in the Task 5 report could be used as a starting place for applying such social and political considerations not used in our development of RSAL values. Some of these criteria are social, political, and economic factors that are outside the scope of our scientific work, yet their impact on the final RSAL value could be significant.

In the following sections, we first list selected RSAL values for all of the scenarios in Table 8 and then provide the detailed probability curves for the key scenarios: DOE-1 (resident). *RAC*-2 (rancher), and *RAC*-2 (child of rancher) scenarios. Based on the results of our calculations, Table 8 lists selected plutonium RSAL values at the 10% probability level; this means there is a 90% probability that the dose limit will *not* be exceeded. This probability level is based on a number of things. First, Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) guidelines, which apply in this case, indicate that the RSAL is intended to assure protection of the "reasonable maximum exposed" (RME) individual, above the 90th percentile of

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the distribution. Additionally, 90% confidence intervals are commonly used in statistical parameter estimation. Considerations such as these directed our decision to select the 10% level as the appropriate level for determining a soil concentration to represent an RSAL.

Table 8: Selected RSAL Values For Plutonium (pCi g^{-1}) at the 10% Probability Level[a]

Scenario	Dose limit[b]	
	15 mrem	85 mrem
DOE-1 (resident)	45	260
DOE-2 (open space user)	Not applied	6600
DOE-3 (office worker)	Not applied	1600
RAC-1 (rancher)	35	Not applied
RAC-2 (child of rancher)	35	Not applied
RAC-3 (infant of rancher)	85	Not applied
RAC-4 (industrial worker)	90	530

[a] At the 10% probability level, there is a 90% probability that the dose limit will *not* be exceeded.
[b] Based on draft EPA guidance from 40 CFR 196. These dose limits were used in the previous DOE/EPA/CDPHE calculations.

The relative importance of pathways for plutonium RSALs depends on the value of the RSAL. The lower RSALs are driven by the occurrence of a fire, which would result in enhanced resuspension and therefore higher air concentrations, which lead to higher inhalation doses. As the importance of inhalation decreased with increasing soil concentrations, other pathways, especially soil ingestion, became more important. In the following discussion, we present the RSAL probability curves for *RAC*'s rancher and child of the rancher scenarios and for DOE/EPA/CDPHE resident scenario and we summarize the key findings about the RSAL probability curve and the dominant exposure pathways for the other scenarios.

- **DOE-1 (resident) scenario:** This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996). The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSALs are about 45 pCi g^{-1} for the 15 mrem dose limit and about 260 pCi g^{-1} at the 85 mrem dose limit. Figure 3 presents the RSAL probability curve for the DOE-1 (resident) scenario resulting from *RAC* calculations.

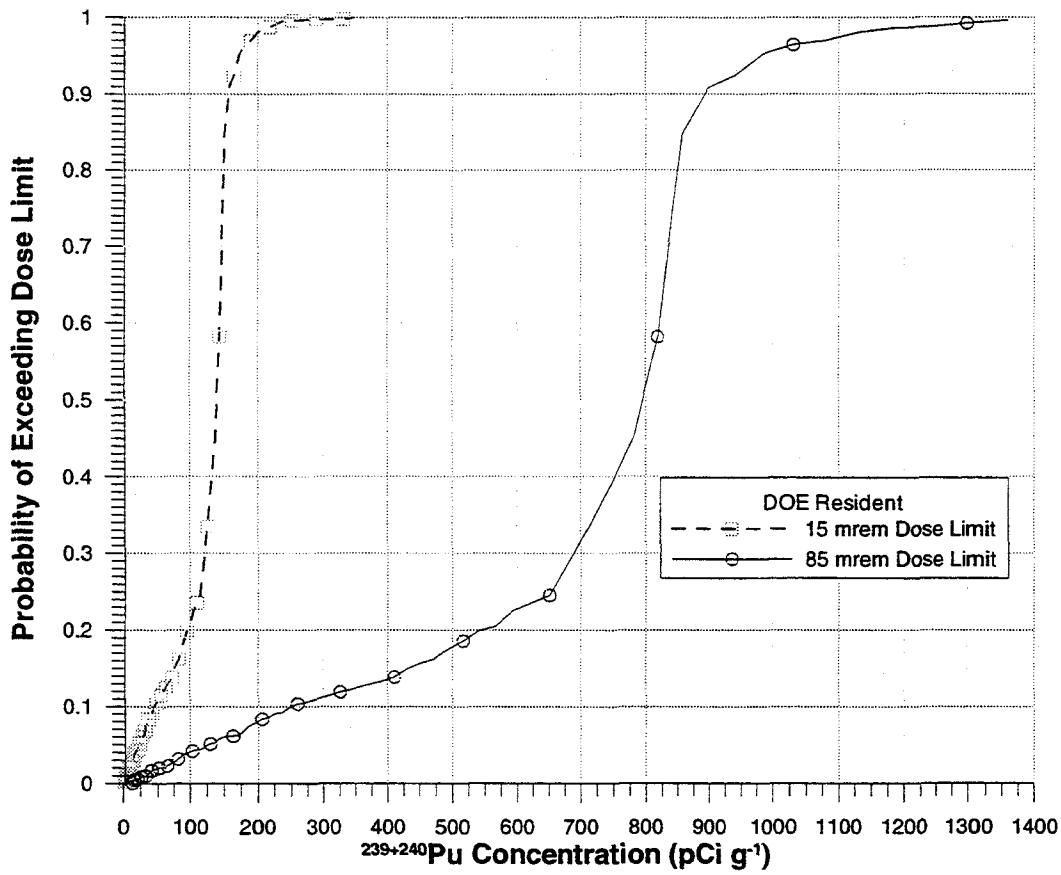

Figure 3. Probability of the total dose exceeding the dose limit for the DOE/EPA/CDPHE resident scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This distribution includes the impact of a fire considered probabilistically.

- **DOE-2 (open space user) scenario:** This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996) and assumes that the site remains as open space and will not be developed in the future. The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSAL for $^{239+240}$Pu, including the sum-of-ratios calculation, was about 6600 pCi g^{-1} for an 85 mrem dose limit.

- **DOE-3 (office worker) scenario:** This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996) and assumes that the site is developed into an industrial park/office complex. The RSALs presented here represent this same scenario calculated stochastically using the using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSAL for $^{239+240}$Pu, including the sum-of-ratios calculation, was about 1600 pCi g^{-1} for an 85 mrem dose limit.

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- ***RAC*-1 (rancher) scenario**: This scenario represents a full-time adult rancher who lives and works on what are now RFETS lands. The probability curve shows two distinct slopes (Figure 4). For ^{239}Pu concentrations less than ~80 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants. For soil concentrations greater than 80 pCi g^{-1}, the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The steep slope of the probability curve for $^{239+240}$Pu concentrations greater than ~80 pCi g^{-1} results from less variability in the doses from the soil and plant ingestion pathways compared to the inhalation pathway. Inhalation doses were proportional to the estimated air concentration, and air concentrations were considerably more variable than soil concentrations. Therefore, RSALs at the 10% probability level (90% probability that the 15 mrem dose limit will not be exceeded) were controlled mainly by the inhalation of resuspended dust. Note that the characteristic inflection point of this probability curve is also seen in the probability curves for the other exposure scenarios. At the 10% probability level, the $^{239+240}$Pu RSAL, including the sum-of-ratios calculation, results in an RSAL of about 35 pCi g^{-1}. Figure 4 presents the RSAL probability curve for *RAC*-1 (rancher) scenario resulting from *RAC* calculations.

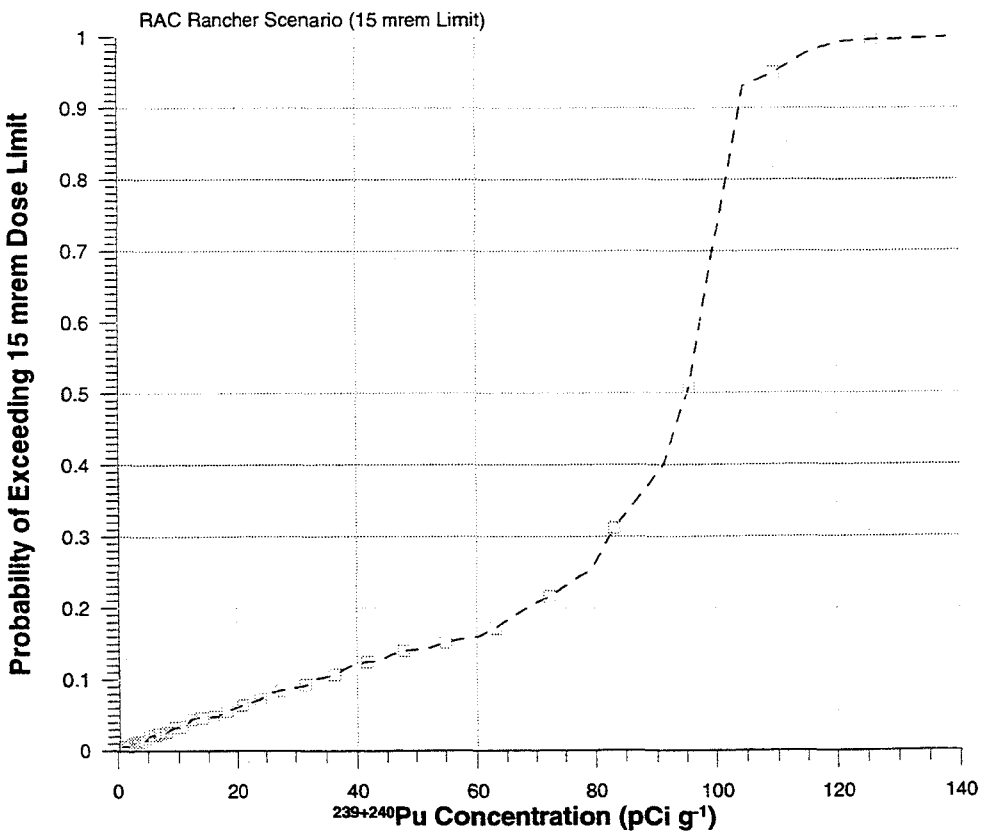

Figure 4. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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- ***RAC*-2 (child of rancher) scenario:** This scenario represents a 10-year old child of a full time resident (rancher) who lives on what are now RFETS lands. The probability curve shows two distinct slopes (Figure 5). For $^{239+240}Pu$ concentrations less than ~60 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants, primarily from fire events. For soil concentrations greater than 60 pCi g^{-1}, the slope of the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The inflection point of this probability curve occurs at a lower $^{239+240}Pu$ soil concentration compared to the adult rancher. Because ingestion rates for the two scenarios were assumed to be the same (75 g y^{-1}), this difference reflects the differences in the ingestion dose conversion factors between the adult and child. At the 10% probability level, the $^{239+240}Pu$ RSAL, including the sum-of-ratios calculation, was about 35 pCi g^{-1}. Figure 5 presents the RSAL probability curve for *RAC*-2 (child of rancher) scenario resulting from *RAC* calculations.

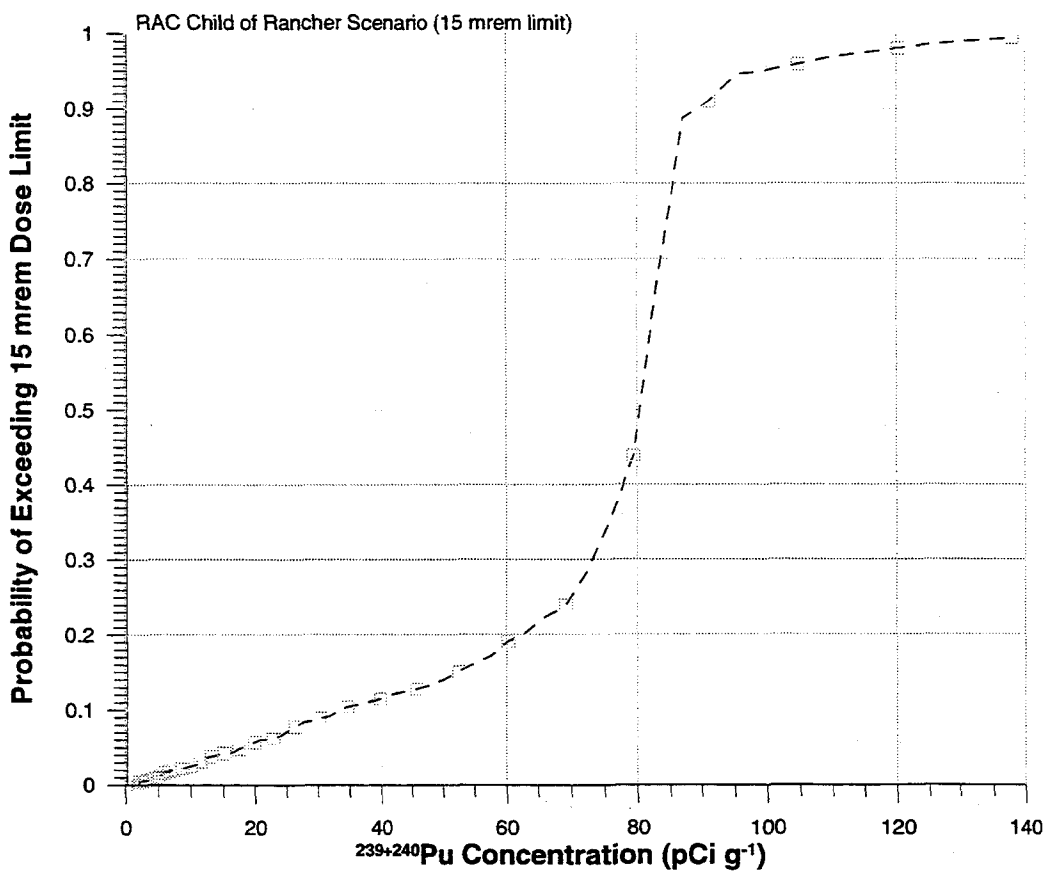

Figure 5. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of the rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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- ***RAC*-3 (infant of rancher) scenario:** This scenario represents an infant of a full-time resident (rancher) who lives on RFETS lands. Like the other scenarios, the probability curve shows two distinct slopes. For $^{239+240}Pu$ concentrations less than ~90 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants. For soil concentrations greater than 90 pCi g^{-1}, the slope of the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The inflection point of this probability curve occurs at a higher $^{239+240}Pu$ soil concentration compared to the adult rancher and child scenarios. This difference reflects the differences in the dose conversion factors and intake rates of contaminated media for the adult, child, and infant. While the dose conversion factors are generally higher for infants, their contaminant intake rates (i.e., breathing rate and food ingestion rates) are generally lower. At the 10% probability level, the $^{239+240}Pu$ RSAL, including the sum-of-ratios calculation, was about 85 pCi g^{-1}.

- ***RAC*-4 (industrial worker) scenario:** This scenario represents an adult who works at an industrial complex at the RFETS. Like the other scenarios, the probability curve shows two distinct slopes. For $^{239+240}Pu$ concentrations less than ~150 pCi g^{-1} (~850 pCi g^{-1} for the 85 mrem dose limit) the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust (plant ingestion was not considered). For soil concentrations greater than 150 pCi g^{-1} (~850 pCi g^{-1} for the 85 mrem dose limit), the probability curve exhibits a steeper slope and is controlled by soil ingestion. The inflection point of this probability curve occurs at a higher $^{239+240}Pu$ soil concentration compared to all other *RAC* scenarios because intake rates of contaminated media are substantially less for this scenario. At the 10% probability level, the $^{239+240}Pu$ RSAL, including the sum-of-ratios calculation, was about 90 pCi g^{-1} at the 15 mrem dose limit and about 525 pCi g^{-1} at the 85 mrem dose limit.

Uranium: Selected Probability Curves and RSAL Values

The previous section described the results of the calculations for the major radionuclides in the soil at Rocky Flats, that is, for ^{241}Am and the several isotopes of plutonium (^{238}Pu through ^{242}Pu). Uranium is also present in the soil at a few locations on the Rocky Flats site in concentrations above natural background, but the history of this contamination is different from that of the americium and plutonium from the 903 Area. For uranium, we assumed fixed isotope ratios for the ^{234}U, ^{235}U, and ^{238}U present at the site and expressed the composite uranium level in terms of a single isotope, ^{238}U. The reported calculations incorporated estimates of parameter uncertainty, and results for each scenario are presented in terms of the probability that the dose limit will not be exceeded. The prairie fire was not considered for the uranium analysis because uranium is more of a hazard when ingested and it was, therefore, of interest to leave it in the surface soil available for leaching into groundwater pathways. Table 9 lists the selected uranium RSAL values at the 10% probability level, again indicating that there is a 90% probability that the dose limit will *not* be exceeded.

Table 9: Selected RSAL Values For Uranium (pCi g^{-1}) at the 10% Probability Level[a]

Scenario	Dose Limit[b]		
	Water pathway on	Water pathway off	
	15 mrem	15 mrem	85 mrem
DOE-1 (resident)		35	200
RAC-1 (rancher)	10	80	
RAC-2 (child of rancher)	10	65	

[a] At the 10% probability level, there is a 90% probability that the dose limit will *not* be exceeded.
[b] Based on EPA guidance from 40 CFR 196. These dose limits were used in the previous DOE/EPA/CDPHE (1996) calculations.

RSALs were presented for uranium isotopes (^{234}U, ^{235}U, and ^{238}U) for three scenarios: the DOE-1 (resident), *RAC*-1 (rancher), and *RAC-2* (child of rancher) scenarios (Killough et al. 2000). The rancher and child of rancher scenarios were chosen because these scenarios yielded the most restrictive RSALs for plutonium. A significant difference between the DOE methodology and our methodology was the area of contamination assigned to uranium. The DOE methodology assumed the area of uranium contamination was the same as plutonium (40,000 m^2). Our investigation indicated that uranium contamination is not as widespread as plutonium contamination and it is mainly limited to past disposal areas or burn pits. Therefore, we treated the uranium contamination as a hot spot and restricted its area to 100 m^2.

- **DOE-1(resident) scenario:** The ^{238}U RSALs at the 10% level (90% probability that the dose limit would *not* be exceeded), were about 35 pCi g^{-1} for the 15 mrem dose limit and about 200 pCi g^{-1} at the 85 mrem dose limit. These RSALs incorporated the sum-of-ratios calculation to include the other uranium isotopes. It is important to point out that the groundwater pathway was treated differently in the *RAC* and DOE/EPA/CDPHE interpretations of this scenario. DOE ignored the groundwater pathway and extracted doses for the year 2000. However, they allowed uranium to be leached from the ground surface at a rate proportional to the background infiltration rate (0.38 m y^{-1}) plus the irrigation rate (1 m y^{-1}). In our calculations, we let RESRAD calculate the maximum dose in the 1000-year time of compliance and extracted RSALs for that time. The time of maximum dose varied between year 2000 and year 2500 depending on the contaminant travel times in the unsaturated and saturated zones. Uranium that migrated to the groundwater was then used for irrigation, thereby contaminating edible plants (direct consumption of water was not considered).

- ***RAC*-1 (rancher) scenario**: Soil action levels were calculated for two cases: one that considered a viable groundwater pathway and the other that assumed all water was derived from offsite sources. Differences between the RSALs with the water pathway on and off were substantial. When the water pathways were turned on, a 1 m y^{-1} irrigation rate was used and resulted in a substantial increase in the removal of radionuclides from surface soil via leaching. However, unlike plutonium, unsaturated zone transit times (the time it takes radionuclides to travel from the contaminated zone to the shallow subsurface aquifer) were typically less than 500 years for uranium isotopes. Consequently, the dose as a function of time typically had two peaks: one at year 2000 (the start time of the simulation) and one after

uranium reached the water well in the aquifer. At the 10% level (a 90% probability that the 15 mrem dose limit will *not* be exceeded), the RSAL for ^{238}U, including the sum-of-ratios calculation, was about 10 pCi g^{-1} with the water pathway on and about 80 pCi g^{-1} with the water pathway off. Doses were dominated by water dependent pathways for ^{238}U RSALs that were <60 pCi g^{-1} with the water pathway on. With the water pathway off, doses were driven by ground exposure and plant ingestion.

- ***RAC*-2 (child of rancher) scenario**: As with the rancher scenario, soil action levels were calculated for two cases: one that considered a viable groundwater pathway, and the other that assumed all water was derived from offsite sources. Again, differences between the RSALs with the water pathway on and off were substantial. At the 10% level (a 90% probability that the 15 mrem dose limit will *not* be exceeded) the RSAL for ^{238}U, including the sum-of-ratios calculation, with the water pathway on was about 10 pCi g^{-1} and about 65 pCi g^{-1} with the water pathway off. With the water pathway on, doses were dominated by water-dependent pathways for ^{238}U RSALs that were <60 pCi g^{-1}. With the water pathway off, doses were driven by ground exposure and plant ingestion.

TASK 6: SOIL SAMPLING PROTOCOL

An important goal of the project was to develop recommendations for a soil sampling protocol for use at the RFETS to obtain soil concentration data for comparison to the soil action levels. Sampling protocols are written descriptions of the detailed procedures to be followed in collecting, packaging, labeling, preserving, transporting, and documenting the samples. Attachment C contains the soil sampling protocol recommendations, reviews existing procedures and protocols for soil sampling, evaluates the quality assurance procedures for sampling, and describes soil sampling protocols in detail based on statistical methods (Thorne and Rood 1999).

Sampling protocols are generally developed using clearly defined guidelines by the EPA and DOE. These guidelines incorporate the iterative data quality objective process and require DOE and its contractor to evaluate several important considerations. These considerations include evaluating sampling and analytical costs in relation to available resources and accepting potential decision errors that may result in remediating sites that are judged contaminated when they are actually below the soil action levels. Conversely, developing a sampling protocol must also incorporate the concerns of the general public and other stakeholders, which are represented by the RSALOP and the soil action level study. Because of the complexity of developing sampling protocols, with the inherent need to balance the concerns of DOE and the RSALOP, developing a comprehensive sampling protocol was not considered possible. Rather, *RAC* recommended elements of a soil sampling protocol considered essential to ensure that representative soil samples are collected for comparison to the soil action levels. These recommendations were provided to the RSALOP for presentation to DOE and its contractor, Kaiser-Hill Company, for incorporation into the soil sampling protocol and procedures to be used for the soil action level process.

RAC conducted a review of the current sampling program used at the RFETS and found that a specific sampling protocol for the soil action levels study had not been developed. However, during this review, several procedures were identified that are available in the Rocky Flats program for incorporation into a sampling protocol. Task 6 also presented recommendations for a

soil sampling protocol to support the final status survey. The final status survey determines the final condition of the site and is performed after decontamination activities are completed. On the other hand, recommendations for a sampling protocol in support of remedial action were not developed for the Task 6 report. Soil sampling in support of remedial action is an important concept; however, a large number of soil samples have already been collected for use in evaluating the nature and extent of contamination in the surface soil at the RFETS (see Attachment C).

RAC provided several recommendations for developing a surface-soil sampling protocol for the final status survey. The following list summarizes some of the recommendations (see Attachment C for the full list). *RAC* recommended the following:

- The data quality objective process should be used to develop the soil sampling protocol for the final status survey.
- DOE should appoint representatives from the RSALOP for inclusion on the data quality objective planning team.
- *RAC*'s technically derived RSAL values from the soil action level probability curves should be used by the RSALOP for comparison to the soil concentration data.
- Profile sampling should be conducted in soil depth increments of 0–3 cm to be consistent with the resuspension model parameters used to develop the soil action levels.
- Soil samples should not be composited; rather, individual soil samples should be analyzed for radionuclide contaminants.
- The arithmetic mean of the soil concentration data and its associated uncertainty at the upper 95% confidence interval should be used for comparison to the soil action levels.
- The non-parametric statistical tests, called MARSSIM, which were developed by the NRC in 1997, should not be used for the soil action level study because these tests compare the median value of the sample distribution to the soil action levels.
- In situ gamma spectroscopy measurement should be performed to identify potential hot spot locations. Hot spots identified by soil samples or in situ gamma spectroscopy measurements should be investigated further to delineate the size of the hot spot and to determine the upper 95% confidence interval of the mean radionuclide concentrations contained in the hot spot.
- DOE should implement an independent verification survey for the RSAL project.

CONCLUSIONS

The primary objective of this project has been to review radionuclide soil action levels (RSALs) adopted by the Department of Energy, the U.S. Environmental Protection Agency, and the Colorado Department of Health and Environment in 1996 for cleanup at the Rocky Flats Environmental Technology Site (DOE/EPA/CDPHE 1996). Another objective has been to recommend a technical approach for independently deriving RSALs for the site. We applied this approach to the Rocky Flats data using the most restrictive exposure scenarios approved by the Oversight Panel and assuming a 10% probability that the 15 mrem per year dose limit will be exceeded (i.e. a 90% probability that the dose limit will <u>not</u> be exceeded). Using this approach, the technically derived RSAL for $^{239+240}Pu$ in soil at Rocky Flats would be 35 pCi g^{-1}. This calculation was corroborated by an alternate method calculation that also resulted in an RSAL at the 10% level of about 37 pCi g^{-1}, suggesting 35 pCi g^{-1} as a technically based RSAL for the

Rocky Flats site. The results as presented are a reasonable indication of RSAL magnitudes based on purely scientific considerations if the prescribed dose is not to be exceeded.

The calculation of uranium RSALs was done somewhat differently than those for plutonium because of significant differences in the nature and extent of contamination and the mobility of uranium in the subsurface. For each uranium scenario, consideration was given to whether groundwater was a viable pathway. A viable groundwater pathway assumed that the surficial aquifer underlying the site would provide enough water for human consumption and irrigation. The impacts of a probabilistic fire were also evaluated but inclusion of this process in our calculations made little difference in the resulting RSALs. Assuming the groundwater pathway was viable and a 10% probability that the dose limit will be exceeded, the technically derived ^{238}U RSAL for the most restrictive scenario (the rancher child) was 10 pCi g^{-1}.

We believe the general approaches and results presented in this report are sound and we recommend their adoption. Data limitations impose uncertainties on estimates of doses, and we have been careful to indicate these uncertainties in our analysis. The project's time and budget goals precluded a more in-depth investigation of several important areas of research that, if addressed in the future, could strengthen this analysis. We have presented these recommendations for further research and recognize that they could change the current results somewhat and improve them as a basis for decision making.

Our methodology is based on several extensions of an earlier approach proposed by DOE/EPA/CDPHE (1996) that used the RESRAD computer program. The contract required that the work consider maximum annual dose limits of 15 and 85 mrem in any year over the next 1000 years. We adopted the 15 mrem per year limit for a technically based RSAL because it is more protective of the public and because our evaluation of risk associated with this dose better corresponds to the target level of risk associated with federal guidance (e.g. CERCLA). Although we considered several computer codes to use as the basis of our analysis, the RESRAD code was adopted because it was the most practical choice and because we were required to make calculations with RESRAD in addition to any other code that may have been selected. Therefore, we designed extensions to RESRAD to include (1) consideration of the heterogeneity of radionuclide concentrations in soil around the site, (2) quantification of the uncertainty in predictions of dose, (3) consideration of additional exposure scenarios, and (4) treatment of the possible occurrence of a large grass fire.

Other factors beyond the scope of this work should be considered in the selection of cleanup strategies for Rocky Flats. The soil action level that is applied for cleanup should be decided by federal and state authorities and the community working together to arrive at a cleanup level that provides long term protection of the public. Figure 6 shows probability curves for the most restrictive scenarios. This figure broadly summarizes the results of our work. Parties involved in the decision process might find the figure useful in their deliberations, keeping in mind the different exposure scenarios represented by the curves and the uncertainties involved.

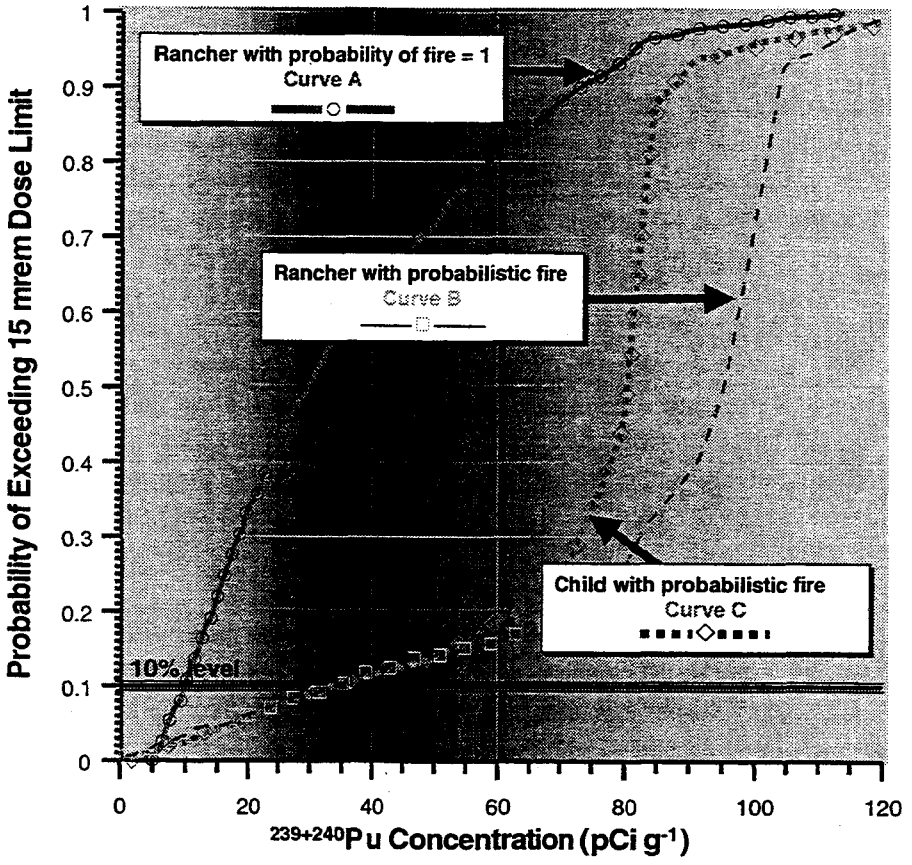

Figure 6. Composite graphic illustrating the most restrictive scenarios and showing a region centered at a soil action level of 35 pCi g^{-1}. Curve A represents the rancher and assumes that a fire occurs with a probability of 1; curve B represents the rancher scenario and takes into account the occurrence of a fire as a probabilistic event; curve C represents the child scenario and, like curve B, incorporates the probability of a fire.

There are several features illustrated in this figure that are important to note. Curve A, defined by the rancher scenario and with the probability of a fire equal to one, likely represents the most conservative set of assumptions and hence the most restrictive radionuclide soil action level. We say "likely" because further research into the impacts of a prairie fire could show that we have underestimated the effects of the fire. Curve B represents the rancher and incorporates a stochastic model of a future fire. With our assumption of a 10% probability of exceeding the dose limit, this curve yields a soil action level of about 35 pCi g^{-1} (the exact value is 33 pCi g^{-1}). Toward the left of the curve, the shape and slope are controlled primarily by inhalation and the probability of occurrence and extent of a fire. However, as the soil concentration of $^{239+240}Pu$ increases, the contribution to dose from ingestion becomes more prominent, and the slope is more influenced by this pathway. Curve C is that of the rancher's child with the stochastic fire model included. This curve is quite similar to that of the rancher with the stochastic fire model but the

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curve indicates this scenario is not as protective as the rancher scenarios in the region of lower RSAL concentrations. At higher RSAL concentrations however, this curve becomes more protective than that of the rancher because the ingestion pathway becomes more influential. The steepness of the curve reflects less uncertainty in the calculation. The rancher scenario with the probabilistic fire is our basis for selecting an RSAL at the 10% probability level.

To give a better visualization of our results, we have underlain Figure 6 with a spectrum that expands in both directions around 35 pCi g^{-1} which is about where the rancher and child of the rancher curves intersect the 10% probability level. Colors are darker near the center of the spectrum and lighter farther out. It is important to understand that curves A, B, and C are based on a sum-of-ratios calculation that incorporates the contribution to dose from other radionuclides present in the soil in addition to $^{239+240}Pu$. The graphic suggests a technically based RSAL of about 35 pCi g^{-1} at the 10% probability level and a range of possible RSALs in both directions centered at this value. Although there is no quantitative basis for the boundaries of this range, it is apparent that going too far in either direction from the center of the spectrum can potentially be problematic for a variety of reasons. Radionuclide soil action levels that are significantly lower may correspond to unrealistically conservative scenario descriptions, which could lead to significantly greater cleanup costs than can be justified. On the other hand, RSALs that are significantly larger lead to a high probability of exceeding the prescribed dose limit and could impact human health. It is especially important to understand that the calculation based on the child scenario and influenced primarily by soil ingestion is scientifically well supported. It is unlikely to change greatly unless values for important parameters change, such as the dose conversion factors or the soil ingestion rate. Therefore, curve C effectively represents an upper bound for the RSAL. If the soil action level were too close to this curve, the probability of exceeding the dose limit is greatly increased.

We also developed an alternate method for calculating acceptable levels of radionuclides in soil. This method was based on calculating annual doses to the receptor for different remediation (i.e., cleanup levels) levels. The remediation level that resulted in a 10% probability that the 15 mrem dose limit would be exceeded defined the RSAL. This method more explicitly addresses the heterogeneity of the site and makes it possible to estimate RSALs that correspond more directly to a remediation strategy than does the sum-of-ratios technique used with RESRAD. The approach is more difficult to implement and therefore has not been fully automated in the analysis. However, because it is more explicit, it is a useful check on the sum-of-ratios method, and we include its results in these conclusions. This alternate calculation resulted in an RSAL at the 10% level of about 37 pCi g^{-1} for $^{239+240}Pu$, suggesting the value of 35 pCi g^{-1} should be strongly considered as a technically based RSAL for the Rocky Flats site.

Our analysis is based on the best available data and methods that we could employ. During the course of our work, we have identified important research that should be completed in order to strengthen our methodology. In addition, changes in the design specifications or scenario assumptions on which this methodology is based would change the results accordingly. This flexibility is quite important to keep in mind because a number of issues that could affect these results have been raised during the course of our work.

While our methodology and the resulting RSAL values are scientifically defensible and are based on sound science, *RAC* believes that additional work could reduce some of the uncertainties and refine the RSALs. There were specific areas where more information or more organized research and scientific inquiry would have allowed us to make better estimates of parameters or

to develop more well-defined methods in our approach. Foremost among these are data that quantify the impact of a prairie fire on the land now occupied by the Rocky Flats site and the data from the Actinide Migration Evaluation studies. Other important areas include:

- effect of prairie fires on the resuspension of material
- time sequence of revegetation following a natural event like a fire
- more realism in the resuspension model for RESRAD
- developing a methodology to estimate the effects of combined exposure to both the uranium hotspots and the widespread plutonium contamination at Rocky Flats
- construction of a computer-implemented model of the Rocky Flats to permit flexibility in analyzing different radionuclides, sources, and pathways
- groundwater transport properties at Rocky Flats
- new discoveries about site-specific distribution coefficients
- potential for accumulation of actinides on offsite lands and water resources
- protection from violation of the Rocky Flats Cleanup Agreement (RFCA) surface water standards for plutonium

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for Rocky Flats site. However, the final decision on setting the RSALs ultimately lies in the hands of the stakeholders, DOE, and other State and federal authorities. There are other criteria that influence the decision-making process for the Rocky Flats site, such as the cost of cleanup, protection of ecological resources, and community values. The approach to cleanup that is ultimately implemented by the DOE at the RFETS will involve many political, social, economic, and moral decisions. It is imperative that all involved in the decision process recognize these factors and the integration of ideas that must go into making a decision of this type.

RAC's task was to evaluate the RSALs adopted for Rocky Flats in 1996, to develop a methodology for independently determining RSALs, and to calculate RSALs for Rocky Flats by applying this methodology. We conclude that applying our method to the exposure scenarios approved by the Oversight Panel, using 15 mrem y^{-1} as a dose limit, and assuming a probability level of 10%, indicates a technically based RSAL for $^{239+240}Pu$ in soil at Rocky Flats of 35 pCi g^{-1}. For uranium, a technically derived RSAL using our methodology and assumptions would be 10 pCi g^{-1}.

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ATTACHMENTS

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APPENDIX A

TASK 1: CLEANUP LEVELS AT OTHER SITES

FINAL REPORT

Task 1: Cleanup Levels at Other Sites

Radionuclide Soil Action Level Oversight Panel

April 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

"Setting the standard in environmental health"

Risk Assessment Corporation
417 Till Road, Neeses, SC 29107
Phone 803.536.4883 Fax 803.534.1995

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FINAL REPORT

Task 1: Cleanup Levels at Other Sites

Radionuclide Soil Action Level Oversight Panel

April 1999

Author
Jill M. Weber, Scientific Consulting, Inc.

Principal Investigator
John E. Till, Ph.D., *Risk Assessment Corporation*

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TASK 1: CLEANUP LEVELS AT OTHER SITES

INTRODUCTION

Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which action should be taken to prevent people from receiving unacceptable radiation doses. 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) have come under scrutiny because of lack of public involvement throughout their development. As a result of public concern, DOE provided funds for the Radionuclide Soil Action Level Oversight Panel (RSALOP) and to hire a contractor to conduct an independent assessment and calculate soil action levels for Rocky Flats. *Risk Assessment Corporation (RAC)* was hired to perform the study. The Rocky Flats Citizen's Advisory Board is administering a grant for the review.

The first task of the study (Task 1: *Cleanup Levels at Other Sites*) was designed to provide the RSALOP with a clear and unbiased evaluation and comparison of previously developed soil action levels for the RFETS and other facilities. This report documents the findings of Task 1.

SOIL ACTION LEVELS AND CONCENTRATIONS AT OTHER SITES

A number of national and international sites have established soil action levels, cleanup criteria, or soil concentrations that are either calculated or measured. These soil action levels have been determined to be protective of human health based on a reasonable land use scenario and predetermined dose criteria. This section briefly summarizes each site in terms of the dose, scenario, and pathways used to calculate the cited soil action level. A later section of the report describes the details of each calculation, including important parameter values, and provides equitable comparisons, where possible.

The one constant across all the sites is that the soil action level was calculated or soil concentration determined for $^{239,240}Pu$. This concentration is provided for each site. Where ^{241}Am soil concentrations are available, they are also given.

The sites evaluated in this analysis are

- Rocky Flats Environmental Technology Site
- Hanford, Washington
- Nevada Test Site
- U.S. Nuclear Regulatory Commission (NRC) codes for remediation
- Johnston Atoll, Marshall Islands
- Enewetak Atoll, Marshall Islands
- Maralinga, Australia
- Semipalatinsk Nuclear Range, Kazakhstan
- Thule, Greenland
- Palomares, Spain.

Rocky Flats Environmental Technology Site

Soil action levels were calculated for the RFETS and documented in a 1996 report (DOE 1996). The RESRAD computer code (Yu et al. 1993) was used to calculate these action levels for three different land use scenarios at two different dose levels.

The three scenarios used in the Rocky Flats calculations were (1) an open space exposure scenario, (2) an office worker exposure scenario, and (3) a hypothetical future resident scenario. Action levels were calculated for ^{241}Am, ^{238}Pu, $^{239,240}Pu$, ^{241}Pu, ^{242}Pu, ^{234}U, ^{235}U, and ^{238}U. Soil action levels for the open space and office worker scenarios were calculated for the annual effective dose equivalent limit of 15 mrem, and the hypothetical future resident scenario soil action levels were calculated for both the 15 mrem and 85 mrem annual effective dose limits, as selected by the DOE (1996).

The open space exposure scenario assumed that an individual visited the area a limited number of times during the year for recreation (DOE 1996). This recreation might include hiking, biking, or wading in creeks. For this exposure scenario, soil ingestion, soil inhalation, and external gamma exposure were the pathways considered. The remaining pathways available in RESRAD (plant ingestion, meat ingestion, milk ingestion, aquatic food ingestion, ground and surface water ingestion, and radon exposure) were not considered (DOE 1996).

The office worker exposure scenario assumed an individual worked mainly indoors, in a building surrounded by paved areas or landscaping. Exposure pathways considered were soil ingestion, soil inhalation, and external gamma exposure (DOE 1996).

The hypothetical future resident scenario assumed that a person resided at Rocky Flats all year and ate produce grown in contaminated soil. Pathways included in this analysis were soil ingestion, plant ingestion, soil inhalation, and external gamma exposure. The pathways removed from consideration were either inconsistent with the site conceptual model or not significant dosimetrically (DOE 1996). For instance, the groundwater and surface water ingestion pathway was removed from the analysis because it was assumed that the water found on the Rocky Flats site would not be sufficient to support domestic use (DOE 1996).

In Table 1, action levels for each scenario (in units of picocuries per gram) are given for each dose level for the radionuclides $^{239,240}Pu$ and ^{241}Am.

Table 1. Soil Action Levels for Each Scenario and Dose at the RFETS (pCi g^{-1})

	Scenario used for soil action level calculation			
Radionuclide	Open Space Exposure Scenario (15 mrem y^{-1})	Office Worker Scenario (15 mrem y^{-1})	Hypothetical Future Resident (15 mrem y^{-1})	Hypothetical Future Resident (85 mrem y^{-1})
$^{239,240}Pu$	9906	1088	252	1429
^{241}Am	1283	209	38	215

These action levels are for single radionuclides. That is, each action level is calculated assuming that the radionuclide of interest is the only radionuclide found on site.

Hanford, Washington

Calculations of soil action levels at Hanford were also done using the RESRAD code, and details of these analyses were published in a 1997 document (WDOH 1997). The two scenarios considered in this study were (1) rural residential exposure and (2) commercial/industrial exposure. These two scenarios are somewhat parallel to the hypothetical resident and office worker Rocky Flats scenarios.

The rural residential scenario assumed a person lived full-time on the Hanford facility. This individual was exposed chronically, indoors and outdoors, to radionuclides in soil, via ingestion, inhalation, and external exposure. The rural residential scenario assumed that the individual worked primarily offsite and engaged in light farming and recreational activities onsite. A portion of the produce, meat, milk, and fish consumed were assumed to come from the site, and drinking water was from an onsite well (WDOH 1997).

The commercial/industrial scenario assumed a person worked onsite, primarily inside a building, although outdoor exposures were also assumed to occur. This scenario assumed that the office worker lived offsite. No ingestion of homegrown food was included in this scenario. Pathways included were limited to external gamma, inhalation of soil, and ingestion of soil (WDOH 1997).

Table 2 shows soil action levels for the two Hanford scenarios, calculated for an annual effective dose limit of 15 mrem.

Table 2. Soil Action Levels for each Scenario and Dose at Hanford (pCi g^{-1})

	Scenario used for soil action level calculation	
Radionuclide	Rural Residential Scenario (15 mrem y^{-1})	Commercial/Industrial Scenario (15 mrem y^{-1})
$^{239,240}Pu$	34	245
^{241}Am	31	210

Nevada Test Site

Calculations of soil action levels were done for the Nevada Test Site by the DOE Nevada Operations Office (DOE-NV 1997). These calculations were performed to show that, subsequent to remediation, the doses received by individuals who may occupy the Tonopah Test Range at the Nevada Test Site would not exceed the dose limits established by the DOE of 100 mrem y^{-1}.

Calculations were done assuming that all areas of the Tonopah Test Range Clean Slate Sites where radiation levels due to $^{239,240}Pu$ exceeded 200 pCi g^{-1} would be remediated to 200 pCi g^{-1} or lower. The RESRAD code was used to calculate dose from the assumed radiation levels in soil.

Four scenarios were used in the dose calculation: a residential rancher, a residential farmer, a rural residence (nonfarming), and a person who worked in light commercial industry. In addition to these adult scenarios, a scenario involving a child who participated in the rancher exposure scenario was included. The rural resident scenario was exposed to external radiation; inhalation of contaminated soil and radon gas and daughter products; and ingestion of soil, drinking water, homegrown produce, meat, and milk. This person was, however, assumed to work offsite and spend only limited time gardening and recreating onsite.

The rancher and farmer scenarios are the closest comparisons to the Rocky Flats rural resident because these scenarios include a significant fraction of time during the year spent onsite. These two scenarios both included exposure pathways of external exposure, inhalation of soil and radon gas and daughter products, and ingestion of soil and drinking water. The rancher scenario included the additional pathways of ingestion of meat and milk, and the farmer scenario included ingestion of homegrown produce. The child scenario implemented the same pathways as the rancher scenario, but it included breathing rates and diet parameters consistent with those of a child.

The industrial worker scenario at the Nevada Test Site is somewhat comparable to the office worker scenario calculated for Rocky Flats. The industrial worker was exposed to external radiation, inhalation of soil and radon, and ingestion of soil and groundwater. This scenario included an 8-hour work day involving both indoor and outdoor work.

Doses for the five scenarios (four adults and one child) were calculated for an achievable $^{239,240}Pu$ soil concentration, determined by the site, of 162 pCi g^{-1}. A soil concentration of 13.2 pCi g^{-1} was presumed for ^{241}Am. Table 3 shows the doses resulting from this soil concentration for both ^{241}Am and $^{239,240}Pu$.

Table 3. Doses for each Scenario for Soil Concentrations Shown at the Nevada Test Site (mrem)

	Scenario used for dose calculation for given soil concentration				
Radionuclide	Rural Residential Scenario	Rancher Scenario	Farmer Scenario	Industrial Worker Scenario	Child Rancher Scenario
^{241}Am (13.2 pCi g^{-1})	1.00	3.56	1.84	0.42	1.61
$^{239,240}Pu$ (162 pCi g^{-1})	10.7	42.6	20.1	3.97	16.7

U.S. Nuclear Regulatory Commission DandD Code Scenarios

The Decontamination and Decommissioning software (DandD) was written for use by NRC licensees to assist them in making screening calculations for cleanup of contaminated facilities. The residential farmer scenario outlined in the DandD code was for a full-time resident of the facility of interest, allowing for some time offsite, as did the Rocky Flats residential calculation. This resident grew as much food as reasonably possible on the facility of interest. The pathways included in the analysis were external gamma exposure; inhalation of soil; and ingestion of soil, water, plants, meat, milk, fish, and poultry. The calculation also included a pathway for irrigation of crops and livestock fodder with contaminated water.

On the whole, the pathway calculations in DandD are highly conservative. We encountered a great deal of difficulty in comparing DandD and RESRAD results because the design of this code is still in preliminary stages and the documentation describing the pathways is not complete or publicly available.

Using default parameters for the DandD residential scenario (Beyeler et al. 1998) (which were selected by the NRC as screening level values), for a soil concentration of 1 pCi g^{-1}, the calculated maximum dose for $^{239,240}Pu$ is shown in Table 4.

Table 4. Dose for Given Soil Concentration in the U.S. NRC DandD Code (mrem)

Radionuclide	Residential Farmer Scenario
$^{239,240}Pu$ (1 pCi g^{-1})	288

Johnston Atoll, Marshall Islands

The dose assessment done for Johnston Atoll in the Marshall Islands was completed after the cleanup efforts were finished. Soil was cleaned to approximately 15 pCi g^{-1} using mining techniques, and this cleanup was verified by Oak Ridge National Laboratory (Wilson-Nichols et al. 1997).

A permissible soil concentration at Johnston Atoll was calculated for a full-time resident exposed to radioactive material through inhalation of contaminated soil. This was the only pathway considered in this dose assessment, and concentrations were calculated for a dose limit of 20 mrem y^{-1}. Because only the inhalation pathway was considered, establishing a detailed scenario was not necessary. Because occupation of the site by the exposed individual is year-around, the Rocky Flats hypothetical future resident scenario exposure traits are the most comparable.

For the Johnston Atoll residential scenario, the dose was calculated for generic compounds of plutonium or americium. The soil concentration was defined as that for $^{239,240}Pu$.

Table 5. Soil Concentration for the Residential Scenario at Johnston Atoll (pCi g^{-1})

Radionuclide	Residential Scenario (20 mrem y^{-1})
$^{239,240}Pu$	17.0

Enewetak Atoll, Marshall Islands

The soil concentrations established for use at Enewetak Atoll have not been discovered to be correlated to a dose assessment. Three different categories of land use were selected, and these categories are shown in Table 6 with their soil concentration limits. Although attempts have been made, the dose calculations associated with these soil concentrations have not been found in the literature.

Table 6. Soil Concentrations Established for Different Land Uses at Enewetak Atoll (pCi g^{-1})

Land use		
Food gathering	Agricultural	Residential
160	80	40

Maralinga, Australia

At the Maralinga Range in Australia, soil concentrations were calculated for a population of semi-traditional aboriginal people permanently residing in the area. Soil concentrations were calculated for a publicly accepted dose limit of 500 mrem. The only pathway considered in this

analysis was exposure via inhalation of contaminated soil. The scenario from the Rocky Flats analysis most comparable to the Maralinga soil concentrations is the hypothetical future resident.

Soil concentrations calculated at 500 mrem for this residential aboriginal population are shown in Table 7.

Table 7. Soil Concentration Calculated for the Residential Scenario at Maralinga (pCi g^{-1})

Radionuclide	Residential Scenario (500 mrem y^{-1})
$^{239,240}Pu$	280

Semipalatinsk Nuclear Range, Kazakhstan

This facility in Kazakhstan was the site of many Russian nuclear tests. The dose and soil concentration information from this facility included no summary of the calculational method used to obtain the dose information. It was not apparent from reading through the available documentation whether the doses and deposited activities were associated with each other in any way. Deposited activities were converted to soil concentrations, assuming normal soil density and depth of contamination. The dose and soil concentration information is shown in Table 8.

Table 8. Activity and Population Dose at Principal Settlements in Semipalatinsk

$^{239,240}Pu$ Deposited Activity (pCi g^{-1})	Individual Dose to Population (mrem)
1.32	Up to 1.5×10^5

Thule, Greenland

The nuclear accident at Thule, Greenland, resulted in concentrations in sediments and not in soils. Because these concentrations are not comparable to Rocky Flats, we do not relate them to Rocky Flats concentrations in this section.

Palomares, Spain

Following a nuclear accident, soil contamination at Palomares, Spain, was immediately cleaned. A dose assessment was completed later by Iranzo et al (1987). For a residential receptor, the pathway of concern was the inhalation of contaminated soil. For this pathway, the acceptable air concentration was calculated based on an annual acceptable dose of 100 mrem. The soil concentration is shown for $^{239,240}Pu$ in Table 9.

Table 9. Soil Concentration for the Residential Scenario at Palomares (pCi g^{-1})

Radionuclide	Residential Scenario (100 mrem y^{-1})
$^{239,240}Pu$	1230

Summary of Available Site Information

Across the mentioned sites, soil concentrations and associated doses vary greatly. The following table is a summary of the soil concentrations measured or calculated at the sites reviewed for this study. Only the scenarios that are comparable to Rocky Flats scenarios are shown. In the next section, we compare all calculations from the different facilities possible to the Rocky Flats in an effort to identify the differences.

Table 10. Soil Concentrations and Associated Doses for ^{241}Am and ^{239,240}Pu Across Sites

Site	Scenario	Soil Concentration (pCi g^{-1})		Dose (mrem y^{-1})	
		^{241}Am	^{239,240}Pu	^{241}Am	^{239,240}Pu
Rocky Flats	Hypothetical future resident	215	1429	85	85
	Office worker	209	1088	15	15
Hanford	Rural resident	31	34	15	15
	Occupational/Industrial worker	210	245	15	15
Nevada Test Site	Rancher	13.2	162	3.56	42.6
	Industrial worker	13.2	162	0.42	3.97
U.S. NRC Codes	Residential farmer	NA	1.0	NA	288
Johnston Atoll	Residential (inhalation)	NA	17.0	NA	20
Enewetak Atoll	Residential	NA	40	NA	unavailable
Maralinga	Residential (inhalation)	NA	280	NA	500
Semipalatinsk	Settlements	NA	1.32	NA	150000
Palomares	Residential (inhalation)	NA	1230	NA	100

SENSITIVITY ANALYSIS

Initial sensitivity analyses of the RESRAD code and parameters used for the Rocky Flats hypothetical future resident scenario at the 85 mrem y^{-1} dose level show that a few parameters dominate the outcome of the action level calculation. These parameters were identified using a single-parameter sensitivity analysis (that is, only one parameter was altered at a time to explore the sensitivity of the RFETS calculation to changes in the parameter). This sensitivity analysis helped identify those parameters that controlled the Rocky Flats soil action level calculation for the Task 1 study. For example, when an action level at another site was significantly different from the RFETS value, we could identify what was likely controlling the difference. Two parameters at the RFETS emerged from the sensitivity analysis as most important and most sensitive to change: mass loading factor and dose conversion factor. The mass loading factor for the RFETS calculations was 0.000026 g m^{-3}. The dose conversion factor for ingestion was 0.000052 mrem pCi^{-1} and for inhalation was 0.308 mrem pCi^{-1}. These dose conversion factors are consistent with Class Y (insoluble) plutonium with a particle size of 1 μm activity median aerodynamic diameter (AMAD). These parameters will be explored in more detail in Tasks 2 and 3, but their importance affects the Task 1 study.

METHOD OF COMPARISON

Action and cleanup levels are often determined independently of dose levels or are based on a dose other than the 15 or 85 mrem y^{-1} used in the RFETS scenario calculations. These varying dose levels made direct comparison more difficult; therefore, we mathematically compared different soil action levels among sites by normalizing the action level to annual dose. In the remainder of this report, annual dose is understood, and dose is represented in units of millirem (mrem). Normalization means that a ratio was calculated for soil action level or concentration to dose level, representing the action level for a unit dose, or 1 mrem. This equitable comparison allows for straightforward identification of pathway, scenario, and parameter differences that affect the ratio. If these differences can be identified among the RFETS and other sites, the ratios between sites should be comparable.

Each ratio is identified in two ways:

1. Dose to soil action level (millirem per picocurie per gram) (mrem $[pCi\ g^{-1}]^{-1}$) and
2. Soil action level to dose (picocurie per gram per millirem) ($[pCi\ g^{-1}]\ mrem^{-1}$).

These ratios are reciprocals. They each have their merits and many different readers find one of the two easier to understand. For a true normalization to dose, the focus should be on the soil action level to dose ratio, which identifies the action level per unit dose, or the soil concentration for each site consistent with a 1 mrem effective dose level. Therefore, if the soil action level to dose ratio is higher for the RFETS than it is for another site, then the allowable soil concentration is greater for the same dose. The opposite situation may also be true. In all cases, this report identifies possible sources for the difference in ratios and calculates the effect of each difference on the ratio to identify the contrast between the ratios.

Because the primary goal of this task was to understand why Rocky Flats soil action levels are consistently greater than those at other sites, we limited out calculations to gaining an understanding of the parameters that drive the action levels to such high levels. Identifying and comparing critical parameters for the RFETS with each site was the endpoint of each investigation. Precisely equating the soil action level to dose ratio between other sites and the RFETS was not our goal. Instead, it was important for us to identify the parameters controlling the action level and show their impact, thereby, making the RFETS action level calculation more transparent.

In some cases, cleanup at a site was conducted independent of dose, and a dose calculation could not be found in the available literature. In these cases, we described the cleanup level along with the soil concentration, but we did not make an effective or meaningful comparison. Without a ratio and some indication of how the calculation was completed, it was impossible to identify the differences among the sites in a way that is meaningful for this study.

COMPARISONS OF ROCKY FLATS SOIL ACTION LEVEL TO SOIL ACTION LEVELS AT OTHER SITES

Several of the previously discussed sites employed alternate action level calculations that lent themselves to comparisons to the Rocky Flats soil action levels for the Task 1 report. These included:

- Hanford, Washington
- Nevada Test Site
- Johnston Atoll, Marshall Islands
- Maralinga, Australia
- Palomares, Spain.

Additionally, the following sections discuss the events that resulted in soil concentrations at Enewetak Atoll, Marshall Islands; Semipalatinsk Nuclear Range, Kazakhstan; and Thule, Greenland. Because no information about dose calculations was available for these facilities, however, our discussion is limited to the facts and does not analyze the calculation or make a comparison of a ratio for these facilities to Rocky Flats. We also describe the U.S. NRC calculations and codes in more detail, but no comparisons of ratios are made to Rocky Flats because of the lack of documentation on the DandD code and the time frame and scope of this project.

Table 11 identifies the dose to soil action level and soil action level to dose ratios for each site where information was available. All ratios shown are for $^{239,240}Pu$, and additional ratios for ^{241}Am are shown when the data were available. The scenarios identified in Table 10 are shown for each site. Ratios and scenarios are described in more detail in the following sections.

Table 11. Ratios for Comparison among Different Sites[a]

Site	Scenario	Soil action level to dose ratio ([pCi g^{-1}] mrem^{-1})		Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)	
		$^{239,240}Pu$	^{241}Am	$^{239,240}Pu$	^{241}Am
Rocky Flats, Colorado	Rural Residential	17	2.5	0.06	0.39
	Office Worker	73	14	0.01	0.07
Hanford, Washington	Rural Residential	2.3	2.1	0.44	0.48
	Industrial Worker	16.3	14	0.06	0.07
Nevada Test Site[b]	Rancher	3.8	3.7	0.26	0.27
	Industrial Worker	41	31	0.02	0.03
Johnston Atoll[c]	Residential (inhalation)	0.85	NA	1.2	NA
Maralinga, Australia	Residential (inhalation)	0.56	NA	1.8	NA
Palomares, Spain	Residential (inhalation)	12.3	NA	0.08	NA

[a] References identified in appropriate section of text.
[b] Ratios from Clean Slate Site 1.
[c] Dose from all alpha particles, soil action level for $^{239,240}Pu$.

It is clear that the values are not the same for all sites. In fact, the soil action level to dose ratio is less than 1 in some cases. For similar scenarios, the Rocky Flats soil action level to dose ratio for $^{239,240}Pu$ is always larger than the ratio at another facility. The following paragraphs provide a site-by-site analysis of each $^{239,240}Pu$ ratio for each scenario and why it differs from the ratio for the RFETS residential or office worker scenario.

Because the ^{241}Am soil action level to dose ratio was either the same for similar scenarios between Rocky Flats and another facility or larger at the other facility, we did not examine ^{241}Am

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further. For this task, we were interested primarily in why Rocky Flats ratios exceeded those at other facilities. This condition did not apply to ^{241}Am.

Hanford, Washington

The Hanford Site in Washington was part of the nuclear weapons production complex and it still operates as a DOE laboratory. Dose reconstruction and cleanup efforts are underway at the facility. As a part the clean up, soil action levels were calculated for the facility using parameter evaluation techniques similar to those undertaken at the RFETS. The Hanford calculation is described in detail in a document issued by the State of Washington (WDOH 1997). All parameter values for Hanford cited and used in this section come from WDOH (1997).

For the residential scenarios at Hanford and RFETS, the soil action level to dose ratio for ^{239,240}Pu at Hanford is 2.3 (pCi g^{-1}) mrem^{-1}, compared to 17 (pCi g^{-1}) mrem^{-1} at Rocky Flats. At Hanford, this scenario represented a person who lived on the current Hanford site all year, eating crops and livestock grown onsite, drinking from site streams, inhaling air, and ingesting soil. The Rocky Flats ratio for plutonium was significantly higher than that at Hanford, so an investigation was warranted.

To compare the Hanford and Rocky Flats ratios, we identified differences in significant parameters and observed how making these parameters the same affected the outcome of the ratio comparison.

The most obvious difference between the Rocky Flats residential scenario and the Hanford residential scenario was the active exposure pathways. The Hanford residential scenario included all exposure pathways allowed in RESRAD except the radon pathway. Compared to Rocky Flats, Hanford included four additional pathways: ingestion of drinking water, ingestion of meat from animals raised on contaminated land, ingestion of milk from animals raised on contaminated land, and ingestion of locally caught fish containing radionuclides.

Holding all other parameters in the Hanford calculation constant, removing these pathways made very little difference to the calculation's outcome. The ratio of soil action level to dose for ^{239,240}Pu changed indistinguishably. It is interesting to note that the ingestion pathways (milk, meat, fish, and drinking water) had almost no effect on the ratio for ^{239,240}Pu. The largest change in soil action level to dose occurred for ^{137}Cs and ^{90}Sr because the transport of these radionuclides is primarily through such food chains. These radionuclides were not of concern for the RFETS, so we focused primarily on changes in the ^{239,240}Pu calculation.

The two parameters identified in the RFETS sensitivity calculation (mass loading factor and dose conversion factor) differed between the RFETS and Hanford calculations. We examined these parameters to see how changes affect the Hanford and RFETS calculations.

A major difference between the Hanford and RFETS calculations was values for dose conversion factors. In the Hanford calculation, dose conversion factors for soluble plutonium were used, which are larger, or more conservative, than those for insoluble plutonium. In the RFETS calculation, plutonium was assumed to be insoluble, and smaller dose conversion factors for both inhalation and ingestion were used. Maintaining our previous pathway modification and using the dose conversion factors for insoluble plutonium in the Hanford calculation, the soil action level to dose ratio for ^{239,240}Pu changed from 2.3 to 9.9 (pCi g^{-1}) mrem^{-1}. This ratio was much closer to the RFETS ratio of 17 (pCi g^{-1}) mrem^{-1}, indicating that the form of plutonium

identified in the environment plays a significant role in the difference between these two calculations.

The mass loading factor used in the Hanford calculation was 0.0001 g m^{-3}, compared to the value used in the RFETS calculation of 0.000026 g m^{-3}. Maintaining all previous modifications to the Hanford calculation and altering the mass loading factor to match the RFETS value, the soil action level to dose ratio for $^{239,240}Pu$ changed from 9.9 to 34 (pCi g^{-1}) $mrem^{-1}$. This large increase in the ratio occurred for two reasons. First, assuming the plutonium was in an insoluble form made inhalation the dominant pathway for dose. Second, decreasing the mass loading factor decreased the amount of plutonium in the air, making less plutonium available for inhalation. The combination of these two changes increased the allowable concentration of plutonium in soil, and correspondingly increased the soil action level for a unit dose.

When the Hanford calculations using RESRAD were run implementing the RFETS pathways and parameter values for mass loading and dose conversion factor, the soil action level to dose ratio for Hanford exceeded that for the RFETS. Table 12 shows the incremental change in the soil action level to dose ratio when the parameters in the Hanford calculation were altered.

For the office worker scenario at Rocky Flats and the industrial worker scenario at Hanford, the pathways analyzed were identical: external gamma exposure, inhalation of soil, and ingestion of soil. The soil action level to dose ratios for $^{239,240}Pu$ for Hanford and RFETS, respectively, were 73 and 16.3 (pCi g^{-1}) $mrem^{-1}$.

We assumed that the same parameter changes that controlled the residential scenario calculation, dose conversion factor and mass loading, would have significant control over this calculation. In fact, this proved to be true. When dose conversion factors were changed to conform to the insoluble form of plutonium, the soil action level to dose ratio for Hanford went from 16.3 to 44. Maintaining this change and changing mass loading from 0.0001 g m^{-3} to 0.000026 g m^{-3}, the soil action level to dose ratio for the Hanford calculation went from 44 to 159 (pCi g^{-1}) $mrem^{-1}$, exceeding the Rocky Flats ratio of 73 (pCi g^{-1}) $mrem^{-1}$. In the case of both residential and worker scenarios, the same parameters controlled the soil action level calculation for $^{239,240}Pu$. Table 12 also shows the changes in parameters that controlled the outcome of the industrial worker scenario.

Table 12. Changes in the Soil Action Level to Dose Ratio with Parameter Value Changes for $^{239,240}Pu$ in the Hanford and RFETS Calculations

Site and Scenario	Parameter change	Soil action level to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats residential	Original calculation	17	0.06
Hanford residential	Original calculation	2.3	0.44
	Remove meat, milk, fish, drinking water	2.3	0.44
	+ change dose conversion factor	9.9	0.10
	+ change mass loading	34	0.03
Rocky Flats office worker	Original calculation	73	0.01
Hanford industrial worker	Original calculation	16.3	0.06
	Change dose conversion factor	44	0.02
	+ change mass loading	159	0.006

Nevada Test Site

The Nevada Test Site was the location of numerous nuclear weapons tests in the 1940s, 1950s, and 1960s during the buildup and testing of the nation's nuclear arsenal. Two documents reported dose calculations for individuals who might live or work onsite after cleanup of the site. One of the dose assessments assumed very realistic scenarios for future site uses and calculations were performed for scenarios including an industrial worker, bomb detonation, removal of safe munitions, aircraft crew flying overhead, ground troops being deployed onsite, explosive ordinance demolition, and a construction worker. In short, these scenarios were designed assuming that the site will be under military control in the future. Ratios associated with these scenarios are large; they are not discussed here because they do not relate to the Rocky Flats scenarios (DOE 1998).

In the second document, doses were assessed for presumed cleanup levels given scenarios similar to those we looked at for the RFETS (DOE-NV 1997). This assessment was performed with RESRAD but reported dose from a given soil concentration, instead of soil action level.

The 100 mrem y^{-1} public dose standard is presumed to be the primary standard for protection of the public based on the DOE Order 5400.5 (DOE-NV 1997). DOE-NV (1997) cited a number of studies detailing soil action levels that resulted in doses similar to or less than this standard. Based upon this information, this dose assessment assumed that the soil needed to be cleaned to a level not exceeding 200 pCi g^{-1} of $^{239,240}Pu$. Given existing concentrations in soils, hypothetical concentrations after remediation were identified, and dose calculations using RESRAD were completed to assess the dose resulting from both the unremediated and remediated soils. If these doses were less than the 100 mrem y^{-1} public limit, the remediation was termed adequate, or even unnecessary, if the pre-cleanup levels met the dose requirement.

Two scenarios from the Nevada Test Site evaluation related most closely to the Rocky Flats scenarios: the rancher scenario and the industrial worker scenario. In the rancher scenario, a person lived on and farmed the land for personal livelihood, eating many of the crops and livestock produced. Pathways included external radiation; inhalation of soil and radon; and ingestion of soil, drinking water, meat, and milk. The same scenario at Rocky Flats did not include radon inhalation, or ingestion of drinking water, milk, or meat. The cited post-remediation soil concentration level for $^{239,240}Pu$ of 162 pCi g^{-1} and dose of 38.9 mrem y^{-1} yielded a soil action level to dose ratio of 3.8 (pCi g^{-1}) $mrem^{-1}$. The ratio for a similar scenario at the RFETS was 17 (pCi g^{-1}) $mrem^{-1}$. Because the plutonium ratio at Rocky Flats was larger than the ratio at Nevada Test Site, this ratio was worthy of examination for this task.

The industrial worker scenario included exposure pathways for external gamma radiation, inhalation of soil, inhalation of radon, ingestion of soil, and ingestion of drinking water. This scenario included two pathways not used in the Rocky Flats calculation: inhalation of radon and ingestion of drinking water. The soil action level to dose ratio for the industrial worker Nevada Test Site calculation for $^{239,240}Pu$ was 41 (pCi g^{-1}) $mrem^{-1}$, compared to the RFETS ratio of 73 (pCi g^{-1}) $mrem^{-1}$. Again, the plutonium ratio was significantly larger.

The primary difference between the RESRAD calculations for the Nevada Test Site and the RFETS was the assumed solubility class of plutonium. The Nevada Test Site calculation used the RESRAD default value for plutonium dose conversion factor, which corresponded to Class W (soluble) plutonium. For purposes of simplicity, changes were made to the readily available RFETS calculation. When dose conversion factors for soluble plutonium were used in the Rocky Flats residential calculation, which originally used Class Y (insoluble) plutonium dose conversion factors, the RFETS soil action level decreased from 1429 to 242 pCi g^{-1}, and the soil action level to dose ratio decreased from 17 to 2.8 (pCi g^{-1}) $mrem^{-1}$.

When this same change was made in the Rocky Flats office worker calculation, the soil action level to dose ratio decreased from 73 to 16 (pCi g^{-1}) $mrem^{-1}$. This single parameter accounts for the difference between these two calculations. Table 13 summarizes the differences between the ratios and the parameter changes employed.

Table 13. Changes in the Soil Action Level to Dose Ratio with Parameter Value Changes for $^{239,240}Pu$ in the Nevada Test Site and RFETS Calculations

Site and scenario	Parameter change	Soil action level to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats residential	Original calculation	17	0.06
	Change dose conversion factor	2.8	0.36
Nevada Test Site residential	Original calculation	4.1	0.24
Rocky Flats office worker	Original calculation	73	0.01
	Change dose conversion factor	16	0.06
Nevada Test Site industrial worker	Original calculation	41	0.02

U.S. Nuclear Regulatory Commission DandD Code Scenarios

The NRC produced its own computer code using models similar to those in RESRAD. This code, called DandD, was designed for use by NRC agencies as a guideline for cleanup and remediation of contaminated sites. Two sets of scenarios were developed for generic use with DandD: (1) scenarios for the release of buildings and (2) scenarios for the release of contaminated land. Only the contaminated land scenarios are comparable to the RFETS calculations. Of the land use scenarios, the residential use, or surface soil, scenario is the most directly comparable to the situation at Rocky Flats.

This scenario assumes residential farming of land with limited gardening activities. The pathways considered are inhalation of soil; ingestion of soil, water, milk, meat, poultry, and fish grown/raised and irrigated by contaminated water; and external gamma exposure. Indoor radon is not considered.

The total effective dose equivalent for the residential scenario for $^{239,240}Pu$, assuming surface soil activity of 1 pCi g^{-1}, is 288 mrem. This yields a soil action level to dose ratio of 0.003 (pCi g^{-1}) $mrem^{-1}$, much smaller than the Rocky Flats ratio.

The differences between these two calculations are numerous, and are not, in all cases, completely transparent without the benefit of the code documentation. Upon running the DandD code, the most noticeable difference is that the primary contributors to the dose are the aquatic pathway (66%), the irrigation pathway (21%), and the drinking water pathway (13%). This results from the use of dose conversion factors that correspond to a soluble class of plutonium, as well as very conservative pathway assumptions relating to concentration factors in fish and plants.

The pathways used in DandD appear to be quite different from those in RESRAD, making it very difficult to compare results from the two without extensive documentation. Representatives from the NRC have indicated to *RAC* that DandD was written for a purpose very different than the calculation of soil action levels, and they did not recommend that actual scenario dose calculations be made with this code; rather, the code is intended to be used for screening level, conservative calculations only.

The differences between the RESRAD and DandD codes are so extensive that a comparison of Rocky Flats residential calculations with RESRAD and the DandD residential farmer scenario is not instructive or possible given the limited time and scope of this project. DandD is reviewed somewhat more extensively in the Task 2 report.

Johnston Atoll, Marshall Islands

Plutonium contamination in the environment at the Johnston Atoll in the Marshall Islands resulted from three accidents in 1962: the destruction of two off-course rockets at high altitude and one explosion on the rocket launching pad (Spreng 1999). Using mining techniques, the soil was cleaned to about 15 pCi g^{-1} (Bramlitt 1988). An independent verification of the cleanup was performed by Oak Ridge National Laboratory (Wilson-Nichols et al. 1997). Currently, a company called GeoCenters is reviewing the cleanup levels and revising the calculations using more realistic receptors. A draft report of this work was due in March 1999 (Spreng 1999).

The scenario used in the Johnston Atoll calculations was a residential scenario using only the inhalation pathway. This resident differed from the Rocky Flats resident in that residence was assumed 24 hours a day, 365 days per year. Using existing information, the soil action level to

dose ratio for a Johnston Atoll resident was calculated to be 0.85 (pCi g^{-1}) $mrem^{-1}$ (Wilson-Nichols et al. 1997). The soil concentration was calculated for doses only from inhaled alpha emitters. The soil screening limit, *SSL*, (or soil action level) was calculated using Equation (1).

$$SSL = \frac{C_{air,acceptable}}{ML \cdot EF} \tag{1}$$

where

$C_{air,\ acceptable}$ = acceptable air concentration (pCi m^{-3})
ML = mass loading (g m^{-3})
EF = enrichment factor (unitless).

The acceptable air concentration is calculated for the accepted annual committed dose. For the Johnston Atoll calculation, the annual committed dose limit was 20 mrem y^{-1}, which corresponds to an air concentration of 2.6×10^{-3} pCi m^{-3} for the alpha emitters, plutonium or americium compounds, assuming a quality factor of 20 (Wilson-Nichols et al. 1997). This air concentration was calculated for Class Y (insoluble) compounds of plutonium that are retained in the lung for years. The committed dose applies to the pulmonary region of the lung.

It is important to note that this calculation was performed based upon a significantly older version of the International Commission on Radiological Protection (ICRP) lung model than that currently in use. The lung model was described in ICRP Publication 19 (ICRP 1972) when recommendations from ICRP 2 (ICRP 1959) were outdated, but ICRP 30 (ICRP 1978) had not yet been published. The ICRP 19 (ICRP 1972) document was prepared by a task group and described an updated version of the lung model. However, ICRP 19 did not yet include calculation of total body dose; the emphasis at this time was still on organ-specific dose. As a result, acceptable air concentrations for the Johnston Atoll were calculated based only on doses to the pulmonary region of the lung. In contrast, the RFETS calculation, which was founded on later ICRP recommendations, describes dose to the entire body. Therefore, the ratios should be compared with caution.

The mass loading factor selected for this calculation was 0.0001 g m^{-3}, as defined by the EPA for developing a soil screening limit (EPA 1977). Even during cleanup and soil disturbance activities at the Johnston Atoll site, mass loading factors were smaller than this value, so the 0.0001 g m^{-3} value was assumed to be a conservatively high (Wilson-Nichols et al. 1997).

The enrichment factor considers how the $^{239,240}Pu$ concentration in the respirable fraction of the soil compares to plutonium concentrations in soil of all particle sizes. An EPA study that looked at five sites in the U.S., including the RFETS, listed enrichment factors for each site (EPA 1977). According to this study, Rocky Flats had the largest enrichment factor of the sites studied across the U.S.. To be conservative, the Johnston Atoll study used an average of the Rocky Flats data to develop an enrichment factor of 1.5.

Using this information and Equation (1), the soil screening limit for the Johnston Atoll was calculated to be 17 pCi g^{-1} for a committed dose equivalent of 20 mrem y^{-1}, giving the ratios cited above. Using Rocky Flats data in this equation helps clarify the differences between the ratios for Johnston Atoll and the ratios for the RFETS.

The first step was to determine the difference between dose conversion factors for the two sites. To extract the Johnston Atoll dose conversion factor from the existing information, we used

an equation for effective dose from inhaled material. Equation (2) calculates dose (in units of millirem) from inhaled material.

$$Dose = V_{inhaled} \cdot C_{air} \cdot DCF \tag{2}$$

where
$V_{inhaled}$ = volume inhaled ($m^3\ y^{-1}$)
C_{air} = concentration in air ($pCi\ m^{-3}$)
DCF = dose conversion factor ($mrem\ pCi^{-1}$).

The volume inhaled in the Johnston Atoll calculation was 8395 $m^3\ y^{-1}$, based on the ICRP reference man (ICRP 1975) for full-time occupation. The concentration in air was 2.6×10^{-3} pCi m^{-3} for a 20 mrem dose. The dose conversion factor that results from inputting these values and rearranging Equation (2) is 0.91 mrem pCi^{-1}. This contrasts with the RFETS dose conversion factor for insoluble plutonium of 0.308 mrem pCi^{-1}. It is important to remember that the RFETS dose conversion factor is for total body dose, and the Johnston Atoll dose conversion factor is only for dose to the pulmonary region of the lung.

Equation (2) can be used to calculate an acceptable air concentration for Johnston Atoll using RFETS parameters. For a Johnston Atoll limit of 20 mrem effective dose limit, RFETS volume inhaled of 7000 $m^3\ y^{-1}$ and RFETS dose conversion factor identified above, the concentration in air is equal to 9.27×10^{-3} pCi m^{-3}.

Equation (1) is used to calculate the Johnston Atoll soil screening limit using Rocky Flats values. The Rocky Flats value for mass loading was 0.000026 g m^{-3}. The air concentration was calculated above, and in the RFETS calculation, no enrichment factor was employed. The soil screening limit for Johnston Atoll using RFETS parameter values is 356 pCi g^{-1}, which gives a soil action level to dose ratio of 17.8 (pCi g^{-1}) $mrem^{-1}$ and matches that of the RFETS. Table 14 summarizes the results of this analysis.

Table 14. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Johnston Atoll and RFETS Calculations

Location	Parameter change	Soil action level to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats	Original calculation	17	0.06
Johnston Atoll	Original calculation	0.85	1.2
	Calculate concentration in air using RFETS dose conversion factor and volume inhaled	3.1	0.32
	+ change to RFETS mass loading	11.9	0.08
	+ change to RFETS enrichment factor	17.8	0.056

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Enewetak Atoll, Marshall Islands

The cleanup levels established for the Enewetak Atoll are very different in scope and intent than those discussed previously. This cleanup was driven more by time, money, and military concerns than an identified limit for concentrations in soil.

The Defense Nuclear Agency published a book describing the cleanup of Enewetak Atoll after numerous U.S. nuclear tests took place there in the 1950s and 1960s (DNA 1981). This book primarily documents the cleanup efforts and decisions made throughout the process; it does not provide a clear assessment of doses and accepted cleanup levels for the islands.

The cleanup of the Marshall Islands was one of the first efforts of its magnitude. Although accidents had occurred at other facilities, guidance was just beginning to be developed for nuclear material soil standards, particularly for transuranics. The EPA guidance on transuranic elements in the environment had not yet been released, and ICRP models for dose were still limited at the time of cleanup.

As a result of limited guidance, decisions about soil cleanup came slowly and only after considerable discussion, disagreement, and finally consensus. As many as three committees produced recommendations for the Enewetak Atoll cleanup, and all committees agreed on some levels and disagreed on others.

The first remediation goal, established by the Environmental Research and Development Agency (ERDA) in conjunction with the U.S. Army Support Command, was to reduce plutonium concentrations in soil to levels below 40 pCi g^{-1}. This concentration level would qualify the land for residential and agricultural use (DNA 1981).

At a workshop held to discuss ERDA plans for the Marshall Islands, doubts and objections to this cleanup strategy were raised, questioning whether the guidelines for soil removal were supportable. As a result of these questions, ERDA convened a panel of scientists, known as the Bair Committee, to review Atomic Energy Commission (AEC) recommendations. An Atomic Energy Commission task group that suggested 400 pCi g^{-1} as an acceptable limit in soil because it was conservatively equivalent to the maximum permissible concentration in air for radiologically unrestricted areas. The task group then introduced a safety margin of a factor of 10, recommending that no cleanup was required below 40 pCi g^{-1}. The areas with soil concentrations between 40 and 400 pCi g^{-1} would be assessed on a case-by-case basis depending on the use of the land. Finally, this task group suggested that after cleanup was initiated, soil levels should be reduced to the lowest possible level (DNA 1981).

Following the AEC recommendations, ERDA established an Operating Plan recognizing that cleanup of all areas to below 40 pCi g^{-1} would require removing large quantities of soil for no appreciable benefit. The Operating Plan suggested conditions for soil use. Condition A specified that an island could be used for food gathering if surface plutonium did not exceed 400 pCi g^{-1}. Condition B allowed agricultural use of land if surface plutonium did not exceed 100 pCi g^{-1}. Residential use, outlined by Condition C, required cleanup to levels below 40 pCi g^{-1}. The final condition involved using the land for all three purposes if the surface conditions met the appropriate requirements and subsurface plutonium concentrations did not exceed 400 pCi g^{-1}.

The Bair Committee approved of the ERDA Operating Plan cleanup criteria and suggested that more specific guidance be established for the soil concentrations between 40 and 400 pCi g^{-1}. When the 1977 EPA guidance on transuranics was released, the Bair Committee adapted its recommendations for agricultural land soil concentrations to 80 pCi g^{-1} and food gathering land

soil concentrations to 160 pCi g^{-1}. These values were apparently based on a dose assessment study performed by Lawrence Livermore National Laboratory. A first study done by Lawrence Livermore National Laboratory was based on the original soil cleanup criteria, but the results were deemed incorrect because of a mathematical error. The Laboratory performed a new dose assessment. Results from this new dose assessment influenced the Bair Committee's decisions concerning action levels for different soil uses.

We could not locate the Lawrence Livermore National Laboratory study in the literature. The Defense Nuclear Agency document lists the radiation doses from this study only unit of millirad; however, these values cannot be converted to effective doses without knowing more about the dose model used to make the calculations. We can assume that Lawrence Livermore National Laboratory scientists used the same model as that used in the Johnston Atoll study, with a large dose conversion factor. However, we would need to have access to the Lawrence Livermore National Laboratory study to make comparisons to RFETS values. We contacted Dr. William Bair, Chair of the Bair Committee, in an attempt to locate documentation. He no longer had copies of the pertinent information, but referred us to Bill Robison at Lawrence Livermore National Laboratory. He has been contacted, and we await a response from him concerning the Lawrence Livermore National Laboratory dose assessment documentation.

Maralinga, Australia

Nuclear weapons trials conducted between 1953 and 1963 by the United Kingdom contaminated the Maralinga site in Australia. This land was the home of semi-traditional Aboriginal tribes, and it became necessary to restore it for their use. A rehabilitation project was undertaken in 1996 because of the extensive $^{239,240}Pu$ contamination in the area. This facility is more difficult to compare to Rocky Flats because RESRAD calculations were not performed. However, a dose evaluation was performed and cleanup criteria were established, so we did have some mechanism to compare the facilities. Doses for the Maralinga facility were calculated for a resident living in a semi-traditional Aboriginal life style, but they focused only on doses from inhalation. This resident lived at the site 24 hours a day, 365 days per year.

In the context of the Maralinga site, the term soil action level is used loosely because cleanup criteria is a more appropriate term. However, we use the term soil action level here for consistency.

The soil action level to dose ratio for the Maralinga site is 0.56 (pCi g^{-1}) $mrem^{-1}$. This ratio was calculated by rearranging the equation used at the Maralinga site to calculate dose. Equation (3) shows the dose calculation used at the Maralinga facility.

$$Dose\,(mrem\,y^{-1}) = C_{air} \cdot BR \cdot DCF \tag{3}$$

where

C_{air} = concentration in air (pCi m^{-3})
BR = breathing rate (m^3 y^{-1})
DCF = dose conversion factor (mrem pCi^{-1})

and

$$C_{air} = C_{soil} \cdot ML \tag{4}$$

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where
C_{soil} = soil concentration (pCi g^{-1})
ML = mass loading (g m^{-3}).

Combining and rearranging Equations (3) and (4) yields Equation (5), which gives a direct calculation of the dose to soil action level ratio. The reciprocal of Equation (5) is the soil action level to dose ratio.

$$\frac{Dose\ (mrem)}{C_{soil}\ (pCi\ g^{-1})} = ML \cdot BR \cdot DCF \tag{5}$$

where all quantities are as previously defined.

The values used in Equation (5) for the Maralinga calculation and the information about the site were extracted from two sources: the journal of *Health Physics* (Johnston et al. 1992) and the Australian Radiation Laboratory (ARL 1998).

Mass loading for the site was determined by simulating some Aboriginal dust raising activities. These data were the only data available to the Australian Radiation Laboratory group, and a value of 0.001 g m^{-3} was used for adults. Breathing rates were taken by the Australian Radiation Laboratory from Haywood (1987). For adults, an annual breathing rate of 8400 m^3 y^{-1} was used. The dose conversion factors were extracted from ICRP 56 (ICRP 1989), but they were corrected for 5 μm AMAD particles because a study indicated this particle size best represented the respirable fraction at the Maralinga site. The dose conversion factor for $^{239,240}Pu$ was calculated assuming the worst case scenario translocation rate for the Australian test sites would be represented by 25% of the plutonium being Class W (soluble) and 75% being Class Y (insoluble). This series of conversions results in a dose conversion factor for $^{239,240}Pu$ of 0.215 mrem pCi^{-1}.

The three parameter values used in Equation (5) lead to a dose to soil action level ratio of 1.8 mrem (pCi $g^{-1})^{-1}$ and a soil action level to dose ratio of 0.56 (pCi g^{-1}) $mrem^{-1}$ for the Maralinga site.

To compare the ratio for the Maralinga site to the Rocky Flats ratio, we inserted RFETS parameter values into the Maralinga calculation. Using the Rocky Flats values for mass loading (0.000026 g m^{-3}), breathing rate (7000 m^3 y^{-1}), and $^{239,240}Pu$ inhalation dose conversion factor (0.308 mrem pCi^{-1}) in Equation (5), yields a dose to soil action level ratio of 0.056 mrem (pCi $g^{-1})^{-1}$ and a soil action level to dose ratio of 17.8 (pCi g^{-1}) $mrem^{-1}$.

Using the Rocky Flats values in Equation (5) accounts for the difference in the two ratios. Table 15 summarizes the changes in the ratios between Maralinga and the RFETS by altering the parameter values used in the calculation.

Table 15. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Maralinga and RFETS Calculations

Location	Parameter change	Soil action level to dose ratio [(pCi g^{-1}) $mrem^{-1}$]	Dose to soil action level ratio [mrem (pCi g^{-1})$^{-1}$]
Rocky Flats	Original calculation	17	0.06
Maralinga	Original calculation	0.56	1.8
	Change to RFETS breathing rate	0.67	1.5
	+ change to RFETS mass loading	26	0.039
	+ change to RFETS dose conversion factor	17.8	0.056

Semipalatinsk Nuclear Range, Kazakhstan

At this location in the former Soviet Union, 124 atmospheric nuclear tests were carried out between 1949 and 1962 (Zeevaert et al. 1997). These tests resulted in environmental contamination and radiation exposure. The contamination was extensively documented and radiation dose rates measured. The results from this work do not yield a soil cleanup level, but they do document existing surface contamination and resulting doses.

It is important to point out that the values given in the literature document either a range of surface radiation levels associated with a single dose or a range of doses associated with a single radiation level. It is very difficult to correlate dose to corresponding soil concentration, not only because surface radiation levels are only tenuously converted to concentrations but also because the surface levels are not related directly to an inhalation dose. Zeevaert et al. (1997) should be carefully reviewed if more information is desired.

For settlements at the Semipalatinsk site, maximum soil activity was given as 11 kBq m^{-2}, corresponding to a soil concentration of 1.32 pCi g^{-1}. We assumed a depth of contamination of 15 cm and a soil density of 1.5 g cm^{-3} to enable us to make this conversion because these factors were not given in Zeevaert et al. (1997). The individual dose to the population resulting from this concentration is identified as 1.5 Sv, or 150,000 mrem. It is not clear from the documentation what this individual dose represents, how it was calculated, or if it correlates in any way to the defined surface soil activity.

The resulting soil concentration to dose ratio is 8.8×10^{-6} (pCi g^{-1}) $mrem^{-1}$. This ratio is fraught with uncertainties, both in measurement techniques and capabilities and difficulty correlating dose to soil concentration in the literature. While this is smaller than the Rocky Flats ratio, it is impossible to account for the differences because the Semipalatinsk soil concentration was measured in the environment, not calculated. Furthermore, Zeevaert et al. (1997) does not describe the dose calculation techniques. We present the ratio only in the interests of completeness, and do not compare it to Rocky Flats.

Another territory affected by the Semipalatinsk tests was Ouglovski, with soil concentrations of 0.66 pCi g^{-1}. The doses cited for this region are external doses, however, and cannot be applied to obtain a ratio.

Thule, Greenland

Near the Air Force Base at Thule, Greenland, on January 21, 1968, a military plane carrying four nuclear weapons crashed and burned. Plutonium contamination was spread about the crash site on the ice, with a maximum contamination level of 14.8 kBq m^{-2}. This site had to be cleaned up before the ice melted in the spring, dictating the time frame of the project. As a result, the only data we have from this crash site are concentrations of plutonium in sediments and estimated dose data from ingestion of sea mussels. Comparisons between this site and the RFETS are impossible because of lack of appropriate data and dissimilar pathway analyses. We report the dose and concentration data in this report for completeness.

After cleanup, the maximum concentration of ^{239}Pu in sediments under the crash site was 1.85 Bq g^{-1}, or 50 pCi g^{-1}. Inhalation is not an appropriate pathway because plutonium is contained in sediments, not dry soil; therefore, the pathway of interest is consumption of mussels. In 1974 (6 years after the accident), the average concentration of plutonium in the edible part of mussels was 0.74 Bg g^{-1} (20 pCi g^{-1}). With a consumption rate of 100 g d^{-1} of mussels for 70 years, the annual committed dose rate to the bone was calculated to be 0.75 mGy (75 mrad) (Church 1998).

Palomares, Spain

Another nuclear accident occurred in Palomares, Spain, on January 17, 1966, when a U.S. Air Force bomber collided with its tanker and exploded above the town. Two of the bomber's four nuclear weapons impacted very near the town and released plutonium. Plutonium oxide contaminated about a 225-hectare (560-acre) area of brushland, farmland, and urban area.

The contamination of this area was so great that immediate cleanup was warranted. Soil concentrations measured just after the accident indicated areas of $^{239,240}Pu$ contamination ranging from 212 μCi g^{-1} (2.12×10^8 pCi g^{-1}) down to 2.12 μCi g^{-1} (2.12×10^6 pCi g^{-1}) (Iranzo et al. 1987). Cleanup was immediately undertaken, with the soil layer at the highest contamination level removed (10 cm deep) and disposed of as radioactive waste. The remainder of the soil was irrigated thoroughly, plowed to a depth of about 30 cm, and homogenized to move contaminated soils to lower levels. At lower levels, the soil would not be available for resuspension to become a potential source of inhalation and dose to residents (Iranzo et al. 1987).

At the time, a dose assessment based on these contamination levels was not performed. The contamination was so widespread that cleanup was the issue at hand. After the cleanup was complete, a monitoring program was established, which included air sampling, soil sampling, crop sampling, and urine and lung counting of the residents.

Air concentrations measured in the environment were compared to (a) annual limits on intake and (b) derived air concentrations from these limits as recommended by the ICRP for radiation workers (ICRP 1978). Because values for acceptable air concentrations for the public were not provided in ICRP 30 (1978), the radiation worker values were multiplied by the ratio of dose limits recommended for the public to those recommended for radiation workers (0.1). This concentration was again reduced to account for ICRP recommendations that effective dose equivalent throughout the life of a member of the exposed population does not exceed the value resulting from a 1 mSv (100 mrem) annual effective dose equivalent. Therefore, acceptable concentration values for members of the public were set at 1.2 mBq m^{-3} (3.2×10^{-2} pCi m^{-3}) for

Class Y (insoluble) compounds of plutonium and 0.5 mBq m^{-3} (1.35×10^{-2} pCi m^{-3}) for Class W (soluble) compounds of plutonium. In the context of the RFETS parameter values, with insoluble Class Y plutonium and a mass loading factor of 0.000026 g m^{-3}, this air concentration corresponds to a soil concentration of 1230 pCi g^{-1}.

Using these values to establish a soil concentration to dose ratio (for the 100 mrem dose for which the air concentration was calculated) results in a ratio for $^{239,240}Pu$ of 12.3 (pCi g^{-1}) $mrem^{-1}$. This ratio is only for inhaled plutonium, and it is based upon the ICRP reference man, who breathes at a rate of 23 m^3 d^{-1} (ICRP 1975). For an exposure time of 8760 h y^{-1} (a full-time resident), this corresponds to an annual breathing rate of 8395 m^3 y^{-1}, which contrasts with the RFETS breathing rate of 7000 m^3 y^{-1}.

Placing the breathing rate of 8395 m^3 y^{-1} into the RFETS calculation yields a soil action level of 1202 pCi g^{-1} and a soil action level to dose ratio of 14.1 (pCi g^{-1}) $mrem^{-1}$. We were unable to discover the reason for the remaining difference between these two ratios during this assessment.

Table 16 summarizes the changes made to the RFETS calculation and ratio.

Table 16. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Palomares and RFETS Calculations

Location	Parameter change	Soil action limit to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats	Original calculation	17	0.06
	Change breathing rate	14.1	0.07
Palomares	Original calculation	12.3	0.08

It is important to note that at the Palomares site, the air concentrations measured in the environment after cleanup were almost always below the acceptable limits, with the exception of four 10-day periods during 1966–1969. During these periods, the increases in contaminated air above the acceptable level could be attributed to cultivation activities, which were hypothesized to raise contaminated soil to the surface and make it available for resuspension (Iranzo et al. 1987).

CONCLUSIONS

The soil action levels at the RFETS are significantly higher than action or cleanup levels at other facilities, even when normalized to dose. However, we understand the reasons for these elevated levels. The outcome of the RESRAD calculation is strongly controlled by a few parameters, and almost without exception, it is these parameters that affect the differences in the soil action levels for a unit dose between sites. The parameters are

- Dose conversion factor (solubility class of plutonium),
- Mass loading (resuspension), and to a lesser degree
- Breathing rate.

Breathing rate is less significant because the range of possible values is limited to within reasonable boundaries. The dose conversion factor varies depending on the assumed solubility of

plutonium. For soluble Class W plutonium, the inhalation dose conversion factor is 0.429 mrem pCi^{-1} and the ingestion dose conversion factor is 0.0035 mrem pCi^{-1}. For insoluble Class Y plutonium, the inhalation dose conversion factor is 0.308 mrem pCi^{-1} and the ingestion dose conversion factor is 0.000052 mrem pCi^{-1} (ICRP 1978). When soluble plutonium is assumed, the ingestion pathway becomes a more dominant contributor to the dose, and the dose per unit intake is considerably greater. For the RFETS, we can determine the appropriate assumption based upon the oxidation state of the plutonium found in the soil at Rocky Flats.

The mass loading parameter can vary over orders of magnitude depending on assumed environmental conditions. Mass loading and similar resuspension parameters have been extensively measured at Rocky Flats under a variety of conditions, and it will be important to use this information to establish a plausible range of values for this parameter. If insoluble plutonium is assumed, inhalation will dominate dose, and mass loading will become a critical parameter.

We reviewed the soil action level to dose ratios for the other sites studied during Task 1 in terms of the calculations, models, and parameters used to calculate soil concentrations and/or dose. In almost every case, differences between sites could be explained by the different assumptions made for one or more of the key parameters identified above (see Table 17).

With Task 1, we identified the input model parameters that are of primary importance in determining the soil action levels so we can carefully review them when completing the Task 3 report, Inputs and Assumptions.

Table 17. Summary of Comparisons between RFETS Calculations and Those for Other Facilities

Location	Parameter change	Soil action limit to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action limit ratio (mrem [pCi g^{-1}}$^{-1}$)
Rocky Flats residential	Original calculation	17	0.06
Hanford residential	Original calculation	2.3	0.44
	Remove meat, milk, fish, and drinking water pathways and change to RFETS dose conversion factor and mass loading	34	0.03
Rocky Flats office worker	Original calculation	73	0.01
Hanford industrial worker	Original calculation	16.3	0.06
	Change dose conversion factor and mass loading	159	0.006
Rocky Flats residential	Original calculation	17	0.06
	Change to Nevada Test Site dose conversion factor	2.8	0.36
Nevada Test Site residential	Original calculation	4.1	0.24
Rocky Flats office worker	Original calculation	73	0.01
	Change dose conversion factor	16	0.06
Nevada Test Site industrial worker	Original calculation	41	0.02
Rocky Flats	Original calculation	17	0.06
Johnston Atoll	Original calculation	0.85	1.2
	Change to RFETS mass loading, enrichment factor, and calculate air concentration using RFETS dose conversion factor and breathing rate	17.8	0.056
Rocky Flats	Original calculation	17	0.06
Maralinga	Original calculation	0.56	1.8
	Change to RFETS mass loading, breathing rate, dose conversion factor	17.8	0.056
Rocky Flats	Original calculation	17	0.06
	Change to Palomares breathing rate	14.1	0.07
Palomares	Original calculation	12.3	0.08

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APPENDIX B

TASK 2: REVIEW COMPUTER MODELS

TASK 3: INPUTS AND ASSUMPTIONS

TASK 5: INDEPENDENT CALCULATIONS OF RADIONUCLIDE SOIL ACTION LEVELS

FINAL REPORT

Task 2: Computer Models

Radionuclide Soil Action Level Oversight Panel

July 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

"Setting the standard in environmental health"

RAC

Risk Assessment Corporation
417 Till Road, Nesses, SC 29107
Phone 803.536.4883 Fax 803.534.1995

FINAL REPORT

Task 2: Computer Models

Radionuclide Soil Action Level Oversight Panel

July 1999

Contributing Authors

George G. Killough, Hendecagon Corporation Inc.
Arthur S. Rood, K-Spar, Inc.
Jill M. Weber, Scientific Consulting, Inc.
Kathleen R. Meyer, Keystone Scientific, Inc.

Principal Investigator

John E. Till, Ph.D., *Risk Assessment Corporation*

Submitted to the Radionuclide Soil Action Level Oversight Panel
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REVIEW OF THE RADIONUCLIDE SOIL ACTION LEVELS AT THE ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE

TASK 2. COMPUTER MODELS

Abstract

This report discusses *Risk Assessment Corporation's* approach to soil action levels (SALs) in context with some computer programs that can be used to calculate them. A mathematical formulation is provided, along with an approach to uncertainty analysis with SALs. Dependence of SALs on exposure scenarios is emphasized. Two sets of scenarios are presented: (1) benchmark scenarios adopted by the Action Levels and Standards Framework for Surface Water, Ground Water and Soils (ALF) Working Group, consisting of members from the Department of Energy (DOE), the Environmental Protection Agency (EPA), the Colorado Department of Public Health and Environment (CDPHE), and Kaiser-Hill; and (2) some refined versions, which are provided for illustration and discussion. Five candidate computer programs were considered for their usefulness in estimating dose and SALs: RESRAD, MEPAS, GENII, MMSOILS, and DandD. RESRAD and GENII tentatively met the requirements set for future computations, which included not only appropriateness of the models implemented, but also the adaptability of the code to command-line execution from a front-end control program. This mode of operation would facilitate customized Monte Carlo analysis, and scripted preprocessing of input data and post-processing of output.

1. INTRODUCTION

This report considers specific computer models and methods that might be useful in the task of setting radionuclide soil action levels (RSALs) for the Rocky Flats Environmental Technology Site (RFETS). The models here reviewed are RESRAD, MEPAS, GENII, MMSOILS, and DandD. They are reviewed for their applicability to this task based on criteria discussed in Section 4. For the purpose of this report, RSALs are defined as radionuclide concentration (activity) levels in a contaminated layer in soil above which remedial action must be taken to prevent people from receiving an annual radiation dose greater than a specified dose limit. The Department of Energy (DOE) has performed calculations of soil action levels with the RESRAD program, which is a DOE product developed specifically for implementing the agency's approach to residual radionuclides in soil (DOE/EPA/CDPHE 1996). A part of the scope of this project is to review these calculations for choice of the parameters that were used in RESRAD, but the review is placed in the larger context of the scientific and technical appropriateness of the models and approach implemented in RESRAD, and whether other programs — or other models and approaches — might be preferred to the one followed by DOE. The parameter choices for RESRAD are a subject of Task 3. The goal of this report is a discussion and comparison of environmental assessment programs that might be used for developing soil action levels for RFETS; as required by the contract, the comparison includes RESRAD.

Before we can discuss the question of suitability of various computer programs for calculating soil action levels, we must make clear our conception of the task to which such programs would be applied. The goal 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. Soil action levels are quantities, one or more per radionuclide, that are computed on the basis of environmental transport models, annual radiation dose limits, and formal assumptions (called exposure scenarios) about the nature and extent of *possible* contact that people *might* have with the site. For a single radionuclide, scenario, and dose limit, the soil action level is that concentration of the radionuclide in the soil that would lead to a maximum predicted annual dose equal to the annual dose limit. For multiple radionuclides, the criterion is more complicated. The concentration of each radionuclide is divided by the respective soil action level, as previously defined. The ratios are summed for all of the radionuclides, and if the sum exceeds 1 for one or more of the exposure scenarios, some action or special attention is indicated. Otherwise (the sum of ratios is less than or equal to 1), the interpretation is that no annual dose limit would be exceeded, and *by that criterion* the radionuclide levels are acceptable. If only one radionuclide is present, the sum of ratios reduces to a single ratio, but the interpretation is the same. Section 2 goes into detail about the definition of soil action levels, the environmental transport models, and the exposure scenarios.

.Our immediate point is that for each radionuclide in the soil, we calculate a quantity called a soil action level, which depends on environmental transport models, annual radiation dose limits, and exposure scenarios. As a matter of common practice, each soil action level is calculated deterministically, which is to say that it represents a single number, typically without indications of uncertainty. Similarly, when the ratios of radionuclide levels divided by soil action levels are summed and compared with 1, the sum of ratios is itself a deterministic quantity, that is, a single number, with typically no indication of uncertainty.

Yet the movement of each radionuclide through environmental media and into possible contact with people is an uncertain process. Although this movement is fundamentally constrained by laws

of physics, chemistry, and biology, models are, of necessity, empirical simplifications of reality, and much of the parametric information on which the models depend is not well known. Contemporary modeling practice explicitly recognizes this state of affairs by treating model parameters and state variables as probability (or uncertainty) distributions, and the calculation propagates the joint uncertainty in the parameters through to the endpoints of the calculation, which, in the case at hand, are the soil action levels and sum of ratios.

When uncertainties in soil action levels are considered, the decision is not so straightforward as in the deterministic case, when the sum of ratios is a single number that is to be compared to 1. When the calculation is stochastic (i.e., takes uncertainties into account), the sum of ratios is a distribution, and one must base a decision on *how probable it is* that the sum exceeds 1. If that probability is small, then one may be willing to forgo action, even though there is some acknowledged possibility that some annual dose limit could be exceeded (indeed, that possibility nearly always exists, even though many conventional calculations do not explicitly recognize it). Section 2.2 goes further into this question. We make the point here, however, that the development and interpretation of soil action levels should follow contemporary methods for incorporating uncertainty into environmental transport modeling. Accordingly, we consider the suitability of various computer programs to provide the necessary machinery.

This report summarizes and compares five prominent computer programs that are configured for environmental assessment: RESRAD, MEPAS, GENII, MMSOILS, and DandD. All of these programs have been developed with support from government agencies, and all have versions that install and execute under Microsoft® Windows 95 or NT. RESRAD, as we mentioned above, is intended to be used in connection with analyzing remediation of radionuclide-contaminated soils at DOE facilities. DOE generally grants access to RESRAD to DOE employees and contractors on DOE-funded projects. MEPAS, which was developed at Pacific Northwest Laboratories (PNL) and is now commercially marketed, is a large multimedia environmental transport program of extensive scope, which is applicable to radioactive and nonradioactive pollutants in many environmental media. GENII, also developed at PNL, is a highly modular radiological assessment system, which provides internal and external dose estimates for exposure through all pathways that are ordinarily considered in environmental radiological assessments. GENII has been under development for more than a decade and is unlikely to be modified further by its developers. MMSOILS, which was developed for the Environmental Protection Agency, is a large multimedia environmental transport program that was designed for screening assessments of chemical contamination. Although it does not treat radioactivity and decay chains, it was included in this review because it could possibly be useful for radionuclides in soils by using stable chemicals as surrogates for radionuclides and performing auxiliary decay-chain calculations external to the program. MMSOILS executables and source code are freely available from an EPA web server. DandD is currently under development by Sandia National Laboratory for the U.S. Nuclear Regulatory Commission (NRC).

We compare these programs with respect to features that are relevant to their possible use in computing soil action levels for the RFETS (Section 4). We draw on documentation distributed with the programs and on published comparisons by authors who participated in the development of the programs (Laniak et al. 1997; Mills et al. 1997). Comparisons of soil action levels developed with some of the programs is the subject of Task 5.

We hesitate to anticipate parameter uncertainties that may be dominant in methodologies for soil action levels until calculations have been done with site-specific data. However, we consider the level of uncertainty associated with the resuspension mechanism to be of sufficient concern that

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it should be raised in this report. This mechanism drives the inhalation exposure pathway and contributes to other pathways (such as deposition on garden vegetables and pasture grass) that could be considered in some scenarios. Models affecting this pathway were changed in RESRAD Version 5.75, although the calculations reported in the soil action levels document (DOE/EPA/CDPHE 1996) were performed with an earlier version of the program. We compare the previous and current versions of the models for this pathway in Section 4.2.3. Predictions of resuspension by the current version tend to be substantially lower than those of pre-5.75 versions.

2. SOIL ACTION LEVELS

Soil action levels may be defined for sites where radionuclides remain in soil at levels that detectably exceed background. Their purpose is to express a possibly complex set of criteria for action that would be taken to protect people who might be exposed to the radioactivity in the near or distant future. Once a set of soil action levels is calculated for the radionuclides of concern, that set may be combined in a sum of ratios with measured or hypothesized concentrations of the radionuclides in soil (each ratio is a soil concentration divided by the corresponding action level) to determine whether the criteria do (or would) call for action, given the measured or hypothesized levels. The soil action levels as defined do not depend explicitly on the actual radionuclide concentrations, because they are determined by using the transport models to calculate levels in soil that would give the limiting annual doses. Thus the same set of soil action levels might be used for determining the need for remediation (based on existing concentrations), planning the remediation (hypothesizing reductions that would result from proposed actions), and verifying that the remediation has been successful (using post-remediation survey results).

The soil action levels depend on four things:

(1) Predicted movement of the radionuclides through environmental media and into potential contact with people (environmental transport models and pathway analysis)

(2) Possible patterns of contact that hypothetical people are assumed to have with the radionuclides in the near or distant future; also, physiological characteristics that would affect the estimation of radiation dose that these hypothetical people would receive (exposure scenarios)

(3) Dosimetric models and data, including radionuclide-specific internal dose coefficients and dose rate factors for external exposure to gamma-emitting radionuclides; these models and data are used to estimate radiation dose to any hypothetical individual with known exposure to radionuclides in the environment (radiation dosimetry)

(4) Annual radiation doses that express protective thresholds for people who might be exposed to the radionuclides (annual dose limits).

The calculation of soil action levels requires environmental transport models (item 1) that consider the various environmental pathways from the source to people who might be exposed (item 2) and methods of radiation dosimetry (item 3) to estimate dose corresponding to the predicted exposure. The purpose is to enable us to see how to control the current levels of the radionuclides in the soil so that the annual radiation dose from these radionuclides to any person who might be exposed to them in ways foreseen in the scenarios (item 2) cannot exceed the annual dose limits (item 4). Section 2.1 presents details of the formulation of the soil action levels.

If the environmental transport models take parameter uncertainties into account, the soil action levels will be represented as a joint probability distribution (the term "joint" indicates possible correlation among the soil action levels), and the sum of ratios (radionuclide concentrations in soil divided by the corresponding soil action levels) is a one-dimensional distribution that must be compared with 1. In this case, we must ask what is the probability that the sum of ratios exceeds 1, and if that probability is acceptably small, one may be willing to accept that exceeding the annual dose limit would be highly unlikely, although possible. Section 2.2 goes into greater detail about uncertainty analysis for soil action levels.

Exposure scenarios are descriptions of characteristics and behaviors of hypothetical individuals who are assumed to have a specified pattern of contact with the radionuclides

originating in the soil at the site. Behaviors would include time regularly spent in one or more locations on or near the site or eating foods from contaminated sources (e.g., a family garden planted in contaminated soil). Characteristics include variables correlated with dose, such as average breathing rates or dietary habits (kg day^{-1} of various food types). Soil action levels may depend on one or more exposure scenarios. Section 2.3 includes additional discussion of scenarios and some examples that may be relevant to the RFETS soil action levels.

The reader is reminded that the validity of soil action levels rests on the information and assumptions that go into their calculation. The calculation anticipates the above-background presence (but not the concentrations) of specific radionuclides and considers only dose limits corresponding to those radionuclides, ignoring any others that may be present. The soil action levels depend on specific exposure scenarios, but the formulation of the scenarios may be quite arbitrary. Thus, it is possible to consider scenarios located in such a way that they would minimize dose from the site and to fail to formulate scenarios based on locations or other assumptions that would tend to maximize dose from the site. Even though the soil action levels do not depend on initial concentrations of the radionuclides of concern, it is recommended that all available information on the spatial distributions of initial radionuclide concentrations be considered as the exposure scenarios are formulated. Otherwise the resulting soil action levels may not impose the desired dose limitation. The implicit nature of soil action levels makes it possible for them to conceal models and assumptions that may not be appropriate for a particular site from users who do not have complete information about the derivation of the soil action levels.

The reader should also be aware that it is always possible, in principle, to avoid soil action levels altogether and to base remediation planning and verification on direct simulations with the data, models, and scenario definitions that would have been used to calculate the soil action levels. That is to say, given a set of measured or hypothesized radionuclide concentrations in soil, the environmental transport and dosimetric models are applied directly to these soil data to estimate annual dose over time to the subjects of the exposure scenarios and thus to determine whether or not dose limitations would be exceeded. Soil action levels need not be calculated at all, and this technique has been employed at various facilities analyzed in Task 1, including Maralinga, Australia, and the Nevada Test Site. This approach has the advantage that its explicit nature draws attention to the numerous elements that go into the estimation of dose as a function of initial concentrations of the radionuclides of concern. Reviewing these models, scenarios, and other data can cause the discovery of errors and assumptions that may not be appropriate for the site under consideration. The disadvantage is some added computational effort, although this disadvantage may have relatively less weight when uncertainties are introduced into the simulations. The current availability and speed of modern computers makes the direct calculation practical for virtually any technical group with the requisite knowledge, whereas decades ago, tables of hazard indices and action levels were essential for decision makers with little or no access to computing equipment that would have made direct computation possible. For example, in the 1960s and 1970s, the International Commission on Radiological Protection (ICRP) published tables of limiting air concentrations for radionuclides in occupational environments, based on dose limitation criteria, whereas contemporary ICRP publications emphasize dose coefficients, on the assumption that any reader has the means to use these coefficients to estimate dose from measured or hypothesized air concentrations of radionuclides.

2.1 Formulation

This section is intended primarily for specialists. It gives mathematical details about the formulation of soil action levels and their relationship to the models and scenarios. The general reader may wish to skip ahead to Section 2.2.

As we shall see in Section 3 and its subsections, it could be desirable to subdivide the RFETS into some number R of subregions, such that the concentration of each radionuclide can be treated as if it were spatially uniform in each subregion. Such a disaggregation would permit an improved representation of so-called hot spots and may offer some advantages in planning and verifying remediation steps. But for the initial discussion of the formulation of soil action levels, we consider a single uniformly contaminated region. At the end of this section, we indicate the more general forms of the formulas when multiple subregions are considered.

It is necessary to define a set of soil action levels for each of the exposure scenarios under study. For any set of radionuclide concentrations $(C_1,\ldots,C_N)$ and scenarios indexed $s=1,\ldots,S$, we can write a sum of ratios for each scenario s as

$$(\mathrm{SR})_s = \sum_{i=1}^{N} \frac{C_i}{(\mathrm{SAL})_{si}}, \quad s=1,\ldots,S \tag{2.1-1}$$

where details of the computation of the denominators are given below. A simple geometric interpretation for $N = 2$ and $S = 1$ is shown in Figure 2.1-1. The $(\mathrm{SAL})_{si}$ will be calculated in such a way that the probability that $(\mathrm{SR})_s \le 1$ is equal to the probability that the dose limit for scenario s is not exceeded. But we must base our soil criterion on the probability that $\max_s (\mathrm{SR})_s \le 1$ (the notation $\max_s(\mathrm{SR})_s$ means the largest of the sums of ratios), so that we control all scenarios by controlling the ones for which potential exposure is maximum. In general, we allow both the numerators and the denominators in the sum in Equation. 2.1-1 to be uncertain quantities. The soil concentrations will come from a joint distribution based either on sampling or existing data. The denominators are based on applicable pathway calculations of dose for the respective scenarios, using Monte Carlo methods to estimate joint distributions. The term "joint" indicates the possibility that there may be correlations among the soil concentrations for different radionuclides, and the denominators may be correlated among scenarios that depend on common pathways (although as a practical matter, we may wish to treat different scenarios as if they were independent). The numerators and denominators will generally be independent.

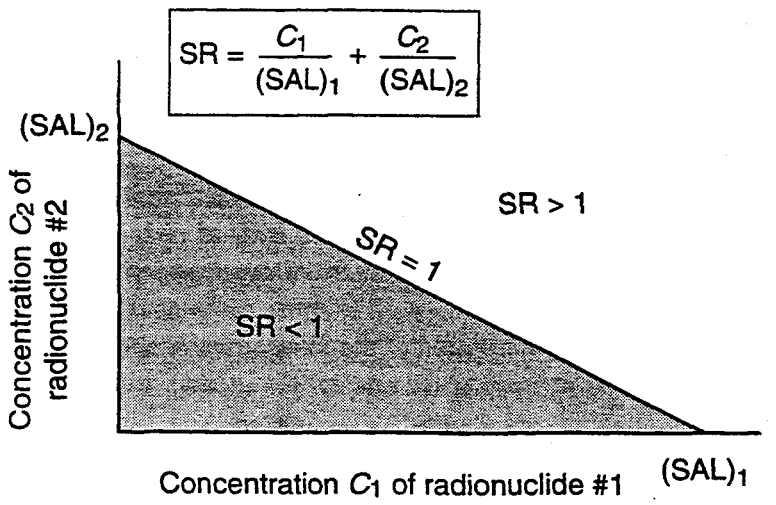

Figure 2.1-1. Geometric interpretation of the sum of ratios (SR) for two radionuclides (N = 2) and one scenario (S = 1). All points (C_1, C_2) on the line represent pairs of concentrations for which the sum of ratios equals 1. For all points in the shaded rectangle beneath the line, the pair of concentrations corresponds to a sum of ratios less than 1 and thus to annual doses that do not exceed the annual dose limit. The concentration pair for any point above the line would lead to an annual dose that exceeds the annual dose limit.

Let us define the transfer function T_{smi} as the quantity that converts a concentration C_i of radionuclide i in the soil to the dose estimate D_{smi}. The subscript s stands for the scenario, and m denotes the particular pathway. The transfer function is something that would be computed by an appropriate environmental transport model. The dose relation for a single radionuclide, scenario, and pathway is

$$D_{smi} = T_{smi} \cdot C_i \,. \tag{2.1-2}$$

Each scenario has a dose limit, and the dose limits are not necessarily the same for all scenarios. Let us denote the limit for scenario s by Δ_s. Then the requirement for the scenario is that

$$\sum_{i=1}^{N}\sum_{m=1}^{M} C_i T_{smi} = \sum_{i=1}^{N} C_i \sum_{m=1}^{M} T_{smi} \le \Delta_s \quad \text{for each } s = 1,\ldots,S \,. \tag{2.1-3}$$

If we divide Eq. 2.1-3 by the dose limit Δ_s and rearrange the second summation, the condition can be expressed as

$$\sum_{i=1}^{N} \frac{C_i}{\Delta_s / \sum_{m=1}^{M} T_{smi}} \le 1, \quad s = 1,\ldots,S, \tag{2.1-4}$$

and this shows us how to define the SALs for the scenarios:

$$(\mathrm{SAL})_{si} = \frac{\Delta_s}{\sum_{m=1}^{M} T_{smi}}, \quad s = 1,\ldots,S, \quad i = 1,\ldots,N \,. \tag{2.1-5}$$

Putting this expression into Equation 2.1-1 defines the scenario-dependent sum of ratios $(\mathrm{SR})_s$. The condition

$$(\mathrm{SR})_s \le 1, \quad s = 1,\ldots,S \tag{2.1-6}$$

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is equivalent to the dose-limitation condition of Eq. 3, in the sense that (2.1-3) holds for each $s = 1,\ldots,S$ if and only if (2.1-6) holds for each $s = 1,\ldots,S$. Thus, to achieve the required dose limitation, we must require that Equation 2.1-6 hold for all s, or equivalently

$$\max_s (\mathrm{SR})_s \le 1 \ . \tag{2.1-7}$$

Of course this requires us to define a separate sum of ratios for each scenario. There is a way to avoid this. We may write

$$(\mathrm{SR})_s = \sum_{i=1}^{N} \frac{C_i}{(\mathrm{SAL})_{si}} \le \sum_{i=1}^{N} \frac{C_i}{\min_s (\mathrm{SAL})_{si}} = (\mathrm{SR}), \tag{2.1-8}$$

where the last equality in Eq. 8 defines a scenario-independent sum of ratios (SR). Now if we impose the condition

$$(\mathrm{SR}) \le 1, \tag{2.1-9}$$

Equation 2.1-9 implies that the inequality of Equation 2.1-7 follows, so that the dose limitation is met for all scenarios. But it does not work the other way, which is to say the following: there may be some sets of soil concentrations for which (2.1-7) would be satisfied but which would violate (2.1-9). Thus (2.1-9) (as defined by (2.1-8)) is a more stringent condition, which could impose lower soil concentrations. Using Equations 2.1-8 and 2.1-9 as the criterion also introduces a complication when we introduce probability and uncertainty.

We regard the C_i and the $(\mathrm{SAL})_{si}$ as uncertain quantities, and consequently we must interpret inequalities like (2.1-3) and (2.1-6) probabilistically. The probability that these equivalent inequalities hold is the probability — based on the uncertainty of the radionuclide concentrations and the environmental transport calculation — that the dose limitation for all scenarios will be collectively met. To estimate this probability, we sample from the joint distribution of the soil concentrations, and from the distributions of the scenario-dependent soil action levels (Equation 2.1-5); using Monte Carlo methods, this permits us to count the number of times during the run the inequality (2.1-4) holds for all scenarios s. Dividing this number by the total number of Monte Carlo cycles gives our estimate of the probability.

If we use criterion (2.1-9) instead, we can estimate the probability that the inequality (2.1-9) holds, but that probability is not the same as the probability that (2.1-7) holds (as we previously pointed out, inequalities (2.1-9) and (2.1-7) are not equivalent: (2.1-9) implies (2.1-7), but not the other way around). The probability of (2.1-7) will in general be larger than the probability of (2.1-9). This approach imposes a more stringent requirement and could require additional remediation to meet the criterion, given the scenarios, the dose limit numbers, and a specified probability that Equation 2.1-9 holds.

As we mentioned at the beginning of this subsection, it could be useful to consider a subdivision of the RFETS into some number R of subregions and to treat soil concentrations of radionuclides as being spatially uniform within any given region (we would hope to avoid this level of complexity). We conclude this section with the more general forms of the equations that define the soil action levels in such a multiple-source environment. We use the indexing variable $r = 1,\ldots,R$ for the subregions ($R = 1$ corresponds to the previous case). For $R > 1$, we have a larger number of soil action levels: whereas in the previous formulation, there were *NS* (one for each radionuclide and scenario), now the number is *NSR* (one for each radionuclide, scenario, and source subregion). We add another index to the concentration $C_i^{(r)}$, and to the transfer function $T_{smi}^{(r)}$, and we define the soil action level as

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$$(\mathrm{SAL})_{si}^{(r)} = \frac{\Delta_s}{\sum_{m=1}^{M} T_{smi}^{(r)}}, \quad i = 1,\ldots,N, s = 1,\ldots,S, r = 1,\ldots,R \tag{2.1-10}$$

and the sum of ratios for scenario s as

$$(\mathrm{SR})_s = \sum_{r=1}^{R}\sum_{i=1}^{N} \frac{C_i^{(r)}}{(\mathrm{SAL})_{si}^{(r)}}, \quad s = 1,\ldots,S. \tag{2.1-11}$$

Using this form of $(\mathrm{SR})_s$, we still apply Equation 2.1-7 as our criterion for dose limitation.

It is important to remember that the compact formulations shown in this subsection conceal a great deal of specific detail about the scenarios and environmental models. We describe a possible set of scenarios in Section 2.3. Sections 3, 3.1, and 3.2 outline a conceptual approach to environmental modeling for the site and the modes of exposure that would be relevant for the site and the scenarios.

2.2 Stochastic SALs

Uncertainty analysis is now regularly applied to environmental modeling. Parametric uncertainty is concerned with the propagation of uncertainty in parameter values through the simulations to the resulting estimates of concentrations in exposure media or to dose or risk. The usual tools are Monte Carlo techniques. In their simplest form, these techniques consist of assigning a probability distribution to each parameter that is treated as uncertain. The simulation is performed a large number of times (usually 1000 if practical), and at the beginning of each repetition, a number is sampled from the distribution associated with each parameter. This random set of parameter values is used to parameterize the model, and the corresponding result (say a dose) is calculated. The 1000 doses define an empirical distribution for the dose quantity. This distribution is considered an estimate of the quantity and represents the propagated uncertainty. Sometimes additional elaboration is necessary, such as the simulation of correlated subsets of the parameters. But the end product is an uncertainty distribution for each calculated quantity.

When the quantities to be calculated are soil action levels, there is no special difficulty in applying uncertainty analysis. The procedure produces an uncertainty distribution for each SAL. Each of these distributions is a marginal distribution of a multivariate joint distribution of the possibly correlated SALs. These correlations need to be preserved for the next step, which is combining the SALs with measured or assumed soil concentrations of the respective radionuclides by forming ratios: soil concentration divided by SAL. The ratios are summed as in the deterministic case, but in the stochastic case there are, say, 1000 sums of ratios, which define an empirical uncertainty distribution of the sum of ratios (SR) quantity. It is this distribution that is compared with 1 to determine the probability that 1 will be exceeded. If, for example, the value 1 occurs at the 95th percentile of the distribution, then the probability that the sum of ratios will exceed 1 is 5%, or one chance in 20. This might be accepted as a small probability of exceeding the dose standard imposed on the scenario from which the SALs were derived. This probability is associated with uncertainties in environmental data and models; it does not come from the scenario itself, which is considered fixed (Section 2.3). If the value 1 occurred at the 60th percentile of the sum of ratios distribution, the probability of exceeding the dose limit would be 40%, which anyone would likely consider large. In that case, some action or attention would be called for. Figure 2.2-1 is a schematic showing two sum of ratios uncertainty distributions corresponding to the two examples we have just given.

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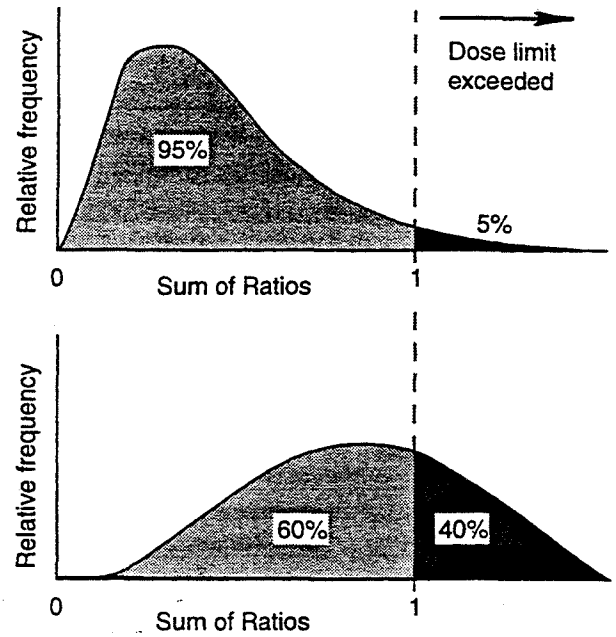

Figure 2.2-1. Schematic illustration of uncertainty distributions for the sum of ratios of soil concentrations divided by the corresponding soil action levels. In the top panel, the probability is 5% that the dose limit for a scenario would be exceeded. In the bottom, the probability is 40%.

2.3 Exposure scenarios

Exposure scenarios describe the characteristics and behaviors of hypothetical individuals who might have some contact with the radionuclides in the soil at the site. The people described by the scenarios live, work, or use the Rocky Flats site for recreational purposes. For the soil action level assessment, a succession of hypothetical individuals over time (for example, 1000 years) is considered. The scenarios represent a means to assess the behavior of radionuclides in the environment in terms of their impact on potentially exposed individuals. A goal for designing the scenarios in this study is that if the hypothetical individuals are protected by specified dose limits, then it is reasonable to assume that others will be protected. The reference scenarios are standards against which levels of radionuclides in the soil at the Rocky Flats site can be measured.

Each scenario represents a single individual with unique physical and behavioral characteristics. These characteristics include variables correlated with dose, such as average breathing rate or dietary habits. Behaviors include time spent indoors and outdoors or eating foods from contaminated sources (e.g. family garden). Exposure scenarios provide assumptions about the nature and extent of possible contact that people might have with the site. Because this study is prospective in nature and has the goal of protecting potentially exposed people from radiation, it may be appropriate to consider biasing some of the scenario parameters in a way that would increase estimated annual dose. However, we recommend that this practice be limited to include only the possible; for example, an individual breathing 24 hours a day at the maximum rate for an Olympic athlete during a strenuous performance is not credible and should not be used to establish an average breathing rate. But it may be appropriate to estimate average breathing rates to include

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periods of strenuous activity, provided the number and lengths of these periods do not exceed what is reasonable.

For the RSAL assessment, some of the parameters are 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. Selecting appropriate parameters for the scenarios depends upon a thorough review of the scientific literature and fully considering the uncertainty (or variability) distributions of the relevant parameters. We use a wide range of references and studies to compile information on parameters. Subsequently, we can generate a distribution of values and sample from the distribution, using Monte Carlo techniques. This process considers the available studies equally. The distributions can be 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 can use a high (or low) 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. Once a parameter value is selected from our distribution of values for use in the scenario, the scenarios are considered fixed just as standards are fixed as a benchmark against which to measure an uncertain value. Behavioral characteristics should be plausible and relevant to the exposure situations and the radiation protection objectives.

Scenarios provide a technical basis for focusing on those pathways and characteristics that are most important in the dose assessment. For example, for plutonium in soils at Rocky Flats, the inhalation pathway will likely prove important. The inhalation or breathing rate affects the transport of airborne contaminants to the respiratory tract and also influences their deposition onto surfaces of the airways and in the pulmonary region. As a result, it is important to exercise care in selecting breathing rate values for each scenario. We have compiled data from numerous published papers to provide perspective in the selection of suitable breathing rates. For soil ingestion, we have reviewed various studies on the unintentional and intentional ingestion of soil by children and adults (e.g., Kimbrough et al. 1984, Calabrese et al. 1990). Simon (1998) developed scenarios based on an extensive review of the literature. The selection of input parameters will be described fully in the Task 3 report for this project. The historic approach for estimating breathing rates over a specified time period is to calculate a time-weighted-average of ventilation rates associated with physical activities of varying time durations. A second approach for determining breathing rates for various populations is based on basal metabolism and measured food-energy intakes and energy expenditures. There is much variability in breathing rates with activity level and age and thus, it is more defensible to use a distribution of values from which to select the input breathing rates (using a high percentile, for example) for an individual scenario.

RAC is evaluating the three scenarios described in the report, *Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement*, dated October 31, 1996 (DOE/EPA/CDPHE 1996), along with additional scenarios that we have proposed and described at the monthly Radionuclide Soil Action Level meetings. *RAC* believes strongly that it is important to describe the process behind the development of the scenarios, to provide the panel with a broad range of scenarios for evaluation, and to consider a number of likely scenarios before final scenarios are selected for the project. In our discussions with the panel, we have used several breathing rate studies as examples of the kinds of data that will be used to develop uncertainty distributions for key parameters. In these meetings, we described the step-wise process to show how breathing rates

can be selected based on activity levels and age, and how these values are summed over a specified time period (e.g. hour, day or year) to yield an annual breathing rate. This demonstration was important to understand that an annual inhalation rate for an airborne radionuclide is based on a weighted average rate, where the weights are determined from the times spent in different activities and at indoor or outdoor locations throughout the day.

We consider the three scenarios outlined in the current Rocky Flats Cleanup Agreement as workable scenarios for the current project. We have designed additional scenarios, too. In some cases we have proposed scenarios with only minor variations from the three current scenarios in the cleanup agreement. For others, we have outlined scenarios with different assumptions about lifestyles and living conditions. Once again, the objective in developing the scenarios is based on the rationale that if the hypothetical individual in the scenario is protected by specified dose limits, then it is reasonable to assume that others will be protected. During the course of designing the exposure scenarios, we had proposed seven additional scenarios. After many discussions with the panel, we focused on four of the proposed scenarios for future RSAL work. The exposure scenarios that are under consideration are described briefly here, beginning with the current Rocky Flats Cleanup Agreements scenarios. Table 2.3-1 summarizes some of the parameter values for those scenarios.

1. The future residential exposure scenario assumes that an individual resides onsite all year and grows and consumes homegrown produce. This person would be exposed to radioactive materials in soils by directly ingesting the soils, by inhaling resuspended soils, by external gamma exposure from contaminated soil and airborne radioactivity, and by ingesting produce grown in contaminated soil. This scenario is from the current Rocky Flats Cleanup Agreement.
2. The open space exposure scenario assumes the person visits the site 25 times per year for recreational purposes, spending 5 hours per visit at the site. The person would be exposed to radioactive materials in the soil by directly ingesting the soils, by inhalation of resuspended soils, and by external gamma exposure from the soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement.
3. The office worker exposure scenario represents an individual who works a 40-hour per week, 50-week per year job indoors in a building complex at the site. It is assumed that this person would be exposed to radioactive material in soils by directly ingesting the soils, by inhaling resuspended soils, and by external gamma exposure from soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement..
4. 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% is spent out of doors. The potential pathways of exposure for this person include inhalation; eating produce from garden irrigated with groundwater, direct soil ingestion from outdoor activities, 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 as described during the monthly RSALs meetings. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
5. Infant in rancher family is 0 to 2 years of age, and onsite 24 hours per day, 365 days per year, or 8760 hr/year. The infant's potential pathways of exposure include inhalation, some direct soil ingestion from outdoor activities, and direct gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
6. The child of the rancher family is assumed to be 5 to 17 years of age, and onsite 24 hours per day, 365 days per year, or 8760 hr/year. The potential pathways of exposure include inhalation, eating produce from garden irrigated with water from a stream on the site, direct soil ingestion, and gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
7. The current onsite industrial worker scenario assumes a person works onsite 8½ 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. *RAC* proposed this scenario for consideration at the February 1999 RSAL meeting.

Table 2.3-1. Summary of Key Scenario Parameter Values for DOE and *RAC* Scenarios

	Current DOE/EPA/CDPHE scenarios			*RAC* recommended scenarios			
				Nonrestrictive		Restrictive	
Parameter	Resident	Open space	Office worker	Current site industrial worker	Resident rancher	Infant of rancher (new-born–2 y)	Child of rancher (5–17 y)
Onsite location				Present industrial area	East of present 903 Area	East of present 903 Area	East of present 903 Area
Time on the site (h d^{-1})				8.5	24	24	24
Time on the site (d y^{-1})				250	365	365	365
Time on the site (h y^{-1})	8400	125	2000	2100	8760	8760	8760
Time indoors onsite (h y^{-1})				900	3500	7740	6600
Time indoors onsite (%)	100	100	100	40	60	90	75
Time outdoors onsite (h y^{-1})	0	0	0	1200	5300	860	2100
Time outdoors onsite (%)	0	0	0	60	40	10	25
Breathing rate (m^3 y^{-1})	7000	175	1660	3700	10000	1900	8600
Soil ingestion (g)	0.2 for 350 d	0.1 per visit for 25 visits per y	0.05 for 250 d	0.20 for 250 d	0.20 for 365 d	0.20 for 365 d	0.20 for 365 d
Soil ingestion (g y^{-1})	70	2.5	12.5	50	75	75	75
Irrigation water source	Ground-water	NA[a]	NA	NA	Ground-water	NA	NA
Irrigation rate (m y^{-1})	1	NA	NA	NA	1	NA	NA
Onsite drinking water source	no	no	no	no	Ground-water	NA	NA
Drinking water ingestion (L d^{-1})	NA	NA	NA	NA	2	NA	NA
Drinking water ingestion (L y^{-1})	NA	NA	NA	NA	730	NA	NA
Fraction of contaminated homegrown produce	1	0	0	0	1	0	1
Fruits, vegetables and and grain consumption (kg y^{-1})	40.1	NA	NA	NA	190	NA	240
Leafy vegetables (kg y^{-1})	2.6	NA	NA	NA	64	NA	42

[a] NA = not applicable.

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3. SITE CONCEPTUAL MODEL

By the term *site conceptual model*, we mean those features of the site that may be explicitly represented by mathematical models for the purpose of predicting dose and deriving soil action levels. The site conceptual model includes the source of the radioactivity, which in this case is the soil on the site with residues of radionuclides that with levels that exceed background by detectable amounts. The model considers the ways in which these radionuclides can deliver dose to people who might come onto the site, and mechanisms by which the radionuclides will move over time from surface soil into other environmental media (environmental pathways), where they may expose people. Thus, the scenarios must be considered part of the site conceptual model, to the extent that they define the receptors and exposure modes (e.g., inhalation, ingestion, or external exposure). The site conceptual model is less detailed than the mathematical models that provide specific formulas for calculating the behavior of the radionuclides over time (dynamic models) and for estimating dose from radionuclide concentrations in environmental media (dosimetric models). It provides a framework within which the mathematical models are organized. Sometimes the term is used to include all parametric information necessary to perform dose calculations. Some of the computer programs that perform the calculations have user-friendly modules that elicit from the operator the information that defines the conceptual site model (RESRAD, MEPAS, GENII). This section gives an overview of the *RAC* conceptual site model for radionuclides in soil at the Rocky Flats site.

Soil action levels are defined in terms of dynamic models that simulate the movement of radionuclide residues in soil through environmental media. They also depend on exposure scenarios, dosimetric models and data, and scenario-specific annual dose limits. The environmental models consider pathways that the radionuclides will follow from the soil to the potentially exposed individuals described by the exposure scenarios. The term *pathway* refers to the succession of environmental media through which the radionuclides move (for example, soil to air, soil to air to garden produce and pasture grass, or soil to surface water runoff to stream). We use the term *exposure mode* for the manner in which the exposure to body organs and tissues occurs. Inhalation, ingestion, and absorption through the skin are modes of intake that lead to exposure from an internally distributed source (internal exposure). External exposure is the result of a person's proximity to a contaminated medium outside the body (air, ground surface, water in which the person swims), such that gamma rays from the radionuclides in the medium deliver dose to the person's organs and tissues. Examples of pathways and corresponding exposure modes are inhalation of radionuclides that are resuspended from the ground surface; ingestion of contaminated soil, either directly or from produce; drinking contaminated surface water (e.g., from a stream that has received runoff from contaminated soil); and consuming animal products (meat or milk) from livestock that have grazed contaminated pasture or drunk contaminated water.

It is important to be as specific as possible about the nature of the models that simulate the movement of the radionuclides along the environmental pathways leading to possible exposure of people. There is no unique approach to the definition of these models: they can range from simple to complicated. The choice of definitions is usually indicated by experience, consideration of the site, and what is mathematically or computationally tractable. Pathways that can be shown to contribute negligibly to the endpoint of the calculation, relative to other pathways, can be omitted, but this must be done with care. Section 3.1 describes the pathways that are potentially relevant to the RFETS. The pathways depend on the exposure scenarios, which we described in Section 2.3. The models, coupled into a system, are treated as uncertain (principally through their parameters:

parametric uncertainty), and when we are given a set of measured or hypothesized concentrations of radionuclides in the soil, we apply Monte Carlo analysis to the sum of ratios to derive a distribution that tells us the probability that the dose limitations will be met.

3.1 Transport pathways

3.1.1 Availability of residual radioactivity in surface soil over time

The behavior of the radionuclides in the surface soil over time is clearly important because of the temporal scope of the scenarios (1000 years). Surface soil with adsorbed radionuclides is entrained into the air by wind action (resuspension) and eventually deposits again on the ground. The processes of resuspension and deposition exist in a quasi steady state cycle, with radioactivity being carried into a region and depositing there and local radioactivity being resuspended and carried away from the region. Over time, this cycle can alter the spatial distribution of radioactivity at the surface. Radioactivity is also removed from the surface soil over time by the action of water, at rates that depend on the amount of precipitation, properties of the soil, and the chemical forms of the radionuclides. Some of the radioactivity moves horizontally (runoff) to streams, and the remainder leaches downward, eventually (except for radioactive decay) crossing the water table and moving into the aquifer. Whatever effect the transport by surface water or groundwater may have on the scenarios that are chosen, it is necessary to take into account the fact that the fraction removed from the surface is no longer available as a source of external exposure or for resuspension. It is important that the transport models deal credibly with this dynamic behavior and persuasively quantify the uncertainties associated with it.

Our approach to multimedia modeling emphasizes the effort to preserve mass balance and to avoid deliberate biasing of environmental concentration estimates. This approach goes hand in hand with our treatment of uncertainty distributions. An example of an approach that would violate this principle is to estimate loss of radioactivity from surface soil by runoff and leaching without accounting for the complementary depletion of radioactivity in the surface soil reservoir. Such calculations can be defended as conservative, but the loss of mass balance accounting generally introduces difficulty into the analysis and interpretation of uncertainty, and we prefer to avoid this difficulty. Our alternative is to try to put the conservatism into the uncertainty distributions, preserving mass balance and minimizing bias. We stress that these are general guidelines, which require interpretation for specific application.

Thus, our conceptual site model treats the soil at any location of interest as a (primarily) vertical reservoir capable of representing distributions of different radionuclide concentrations over time. The model considers variable partitioning of each radionuclide into an aqueous (dissolved) and an adsorbed (adhering to soil) component. The first component moves with water that infiltrates the soil; the latter component is attached to soil matrix and mobile particles. Material attached to the soil moves by (1) surface weathering of the soil and (2) transferring from adsorbed to aqueous state when unsaturated water infiltrates the vadose zone. Radioactive ions also move from the aqueous state to attach to available sites on the soil matrix. The partitioning is usually characterized by a coefficient written as K_d, with units (mL g^{-1}). In environmental work, K_d is interpreted as the ratio at steady state of the radionuclide activity adsorbed on soil divided by the radionuclide activity remaining in solution. However, the steady state assumption is sometimes questionable in the interpretation of process modeling. Narrower definitions of K_d are used in laboratory work, and criticisms of environmental soil modeling often turn on the use of this parameter and its different interpretations (Jirka et al. 1983).

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We also need to mention the mechanism of colloidal transport, in which ions of the radionuclide attach to mobile submicron particles (colloids), which move by the action of water through intersticial spaces in soil and aquifers (Honeyman 1999). Recent investigations at the Nevada Test Site confirmed colloidal transport of $^{239+240}Pu$ a distance of 1.3 km in groundwater. The ^{240}Pu:^{239}Pu ratio of the sample fingerprinted a particular underground nuclear test as the origin of the displaced plutonium (Kersting et al., 1999). The high affinity of plutonium for attachment to rocks has long supported assumptions of low mobility in predicting the movement of plutonium in soil and groundwater, but the introduction of colloidal transport models may eventually alter this pattern. No such explicit mechanism is included in any of the computer programs discussed in this report, and indeed, there is as yet no body of data that could credibly calibrate models of colloidal transport for the Rocky Flats site.

Given the initial amounts of radionuclides in the surface soil, the model predicts the evolving vertical distribution over time as the radioactivity is redistributed by the processes described above. At any subsequent time it is possible (in principle) to evaluate the predicted concentration in soil near the surface that would be available for resuspension, uptake through the roots of plants, direct ingestion, or exposing people to gamma rays from this external source. Not all computer programs handle the removal and redistribution mechanisms in the same way, and the results may differ.

3.1.2 Spatial disaggregation of soil

Contamination of the Rocky Flats reservation by some of the radionuclides of concern is far from uniform. Figure 3.1.2-1 shows the variation of ^{239}Pu concentrations along a transect eastward from the 903 Area, plotted from data of Webb (1996). Litaor et al. (1995) show contour plots of $^{239+240}Pu$ concentrations in the soil. Programs such as RESRAD proceed on the assumption of a uniformly contaminated area (subject to variation within a factor of 3). For some scenarios it could be desirable to subdivide the site area into some number P of plots, each of which can be treated as having a uniform concentration of each radionuclide, but with concentrations varying from one plot to another. Such subdivision might be of assistance in the planning for remediation, because the effects of reducing the most contaminated plots by various amounts can be studied explicitly. However, given the relatively small area of the most highly contaminated soil, we would be reluctant to recommend this refinement without careful evaluation of any factors that might seem to indicate it. We have included equations for area disaggregation near the end of Section 2.1 for the sake of completeness.

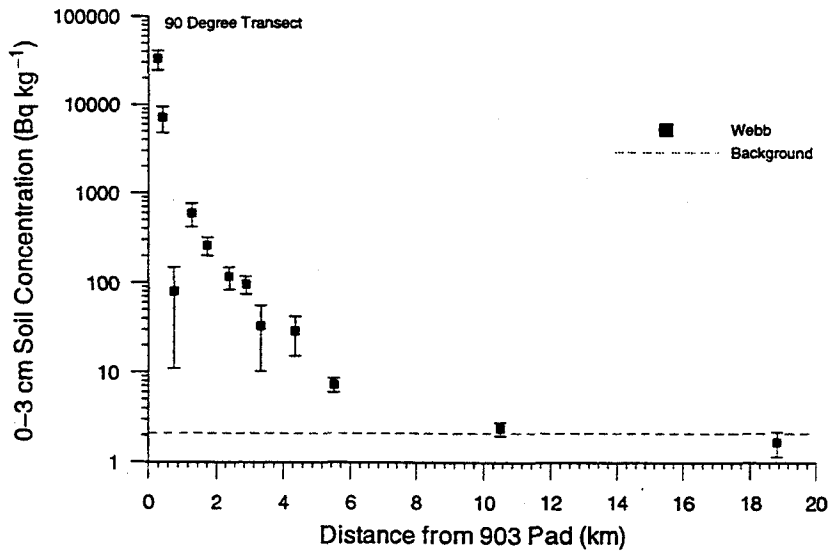

Figure 3.1.2-1. Plutonium-239 concentrations in soil (Bq kg^{-1}) at RFETS along a 90° transect (eastward) from the 903 Pad area. The data are from Webb (1996).

3.1.3 Resuspended contaminated soil

The experience of *RAC* in the Rocky Flats Dose Reconstruction project indicates that the inhalation of resuspended soil that was contaminated by plutonium from the 903 Pad is a potentially significant exposure pathway. Its importance depends on how the scenarios are defined, primarily with respect to location relative to the locations of highest contamination of $^{239+240}Pu$. In Section 2.3, we described a possible scenario that assumes eventual loss of institutional control of the site and that families establish homesteads west of Indiana Street, within the area most affected by the 903 Pad. Such a location (within the contour marked 10 Bq kg^{-1}) would maximize the inhalation exposure to resuspended plutonium, given the prevailing westerly winds, whereas locations west of the RFETS near Highway 93 would correspond to lower inhalation doses. It seems clear that this exposure pathway must be considered, whatever the decisions about scenarios might be.

A serious problem in dealing with any exposure pathway that depends on resuspended soil is the uncertainty introduced into the calculation by the inexact characterization of the mechanisms. Resuspension occurs as a result of wind action on available soil particles, at a rate that depends on wind speed, gross characteristics of the ground surface (roughness of the soil, vegetation, and other objects), and characteristics of the soil, such as size distributions of the particles and tendency of the soil to form less-erodible crusts. The resulting air concentration (which determines exposure by inhalation and external exposure to gamma rays from the diffused particles) depends not only on the resuspension rate but also on stability parameters for the atmosphere, which establish a vertical profile of concentration, and on the deposition rate at which the airborne particles return to the ground. Local levels of contamination borne by the resuspended particles are diluted by particles that entered the air at various distances upwind from the contaminated site. The complexity of this environmental system guarantees large uncertainties in predictions of process-level models for which parameters are difficult or impossible to quantify by direct measurements. (We use the term *process-level* to refer to models that are formulated in terms of the processes of fundamental

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physics, chemistry, and biology, as opposed to *empirical* models, which may summarize many complicated processes in a few directly measurable parameters. This is an oversimplification since most models are empirical at some level, but the distinction is sufficient for this discussion.)

Langer (1986) reports measurements of airborne ^{239}Pu and airborne dust at heights of 1, 3, and 10 m from November 1982 through December 1984 (measurements at 3 m covered a shorter period). The dust-collection and wind-measurement apparatus was placed 100 m southeast of the former East Gate of the plant, near the 903 Pad, and less-detailed measurements of airborne ^{239}Pu were also taken from three samplers near the former East Gate. Both the dust and radioactivity measurements give a crude indication of particle size distributions. A relatively long record of this kind provides what may be the most useful information for calibrating empirical models of resuspension from the field east of the 903 Pad, although this information is still very limited and must be applied with care. But these measurements do provide long-term averages of ^{239}Pu air concentrations that likely approach the maximum for the site. These measurements implicitly take into account the dilution from upwind dust of low contamination, whereas modeling this dilution is a highly uncertain exercise. Krey et al. (1976) used air and soil sampling data from three sites in the field east of the 903 Pad to estimate that only 2.5% of the respirable dust came from local resuspension. This result cannot be considered generically applicable because of uncharacteristically high precipitation during the sampling period, but it does illustrate the point.

The computer programs under investigation approach the resuspension mechanism in one of three ways (in some cases, the user is offered an option of more than one method). (1) *Mass loading*, in which a measured or hypothesized concentration of airborne dust (g m^{-3}) is multiplied by the local concentration of radionuclide on resuspendable soil particles (Bq g^{-1}) to produce an estimate of airborne radioactivity concentration (Bq m^{-3}). (2) *Resuspension rate* (m^{-2} s^{-1}), which may be estimated as the air concentration of dust at a reference height (g m^{-3}) times an average deposition velocity (m s^{-1}) divided by the mass of resuspendable particles per unit area (g m^{-2}). (3) *Resuspension factor*, which may be defined as the air concentration of dust at a reference height (g m^{-3}) divided by the mass of resuspendable particles per unit area (g m^{-2}). The resuspension factor has units m^{-1} (or g m^{-3} airborne per g m^{-2} of resuspendable soil particles) and is equal to the resuspension rate divided by the average deposition velocity. These three approaches to resuspension modeling must be handled with some care. Used without adjustment, they incorporate a tacit assumption that the calculated air concentration of radioactivity-bearing dust is undiluted by uncontaminated dust from upwind. The resuspension factor, for example, is interpreted as the air concentration of dust per unit areal mass of resuspendable particles. This very definition tempts one to impute the local air concentration entirely to the local supply of available particles. But under the usual windy conditions, this assumption would be approximately valid only for large uniform areas upwind from the reference location, and the same is true when the particles are assumed to be contaminated with radioactivity.

All three of these approaches require quantification from the analyst or from default values or formulas supplied by the programs. In this respect, the mass loading approach is perhaps the most direct, requiring as its parameter the very air dust concentration that we seek to estimate. The parameter estimate should be based on measurements taken at the site and averaged over as long a period as possible. The measurements of Langer (1986) indicate a mean total dust concentration of 47 μg m^{-3} with standard deviation 9.0 μg m^{-3} at the 1-m height for the period November 1982 through December 1984. This total quantity, however, includes a substantial fraction of particulate mass in a size range that is not regarded as respirable (59%). If the coarsest category of particles is

discarded, the mean concentration is only 19.2 μg m^{-3}. Most of the resuspended plutonium activity (81%) at the 1-m level is associated with the coarse (non-respirable) particles, leaving only 19% associated with respirable particles. We cite these data to illustrate the point that one should consider the question of the size distribution of the airborne dust and the distribution of plutonium activity over the airborne particles in order to make credible estimates of inhalation dose. The computer programs that implement mass loading do not exercise this judgment, although default values of some parameters may be supplied. Another complication is that air samplers lose efficiency as the particle aerodynamic diameter increases, and the efficiency loss is aggravated by the high wind events that cause much of the resuspension. Thus the measurements taken at Rocky Flats are subject to uncertainties of interpretation, and these uncertainties need to be quantified and incorporated into the calculation.

An approach to resuspension rate estimation is given by Cowherd et al. (1985) in an EPA report. Equations are provided for wind-driven resuspension associated with infinite and limited reservoirs of resuspendable particles. The parameterizations for the EPA models are given in detail, with instructions for coarse particle-size measurements in the field. The report also treats resuspension by mechanical means, such as vehicular traffic. The methods presented are intended to provide a "first-cut, order-of-magnitude estimate of the potential extent of atmospheric contamination and exposure resulting from a waste site or chemical spill, within the 24-hour emergency response time frame." Variants of these models are incorporated into MEPAS, with the necessary graphs and figures from Cowherd et al. (1985) reproduced in the MEPAS documentation. But by use of the front-end technique described in Section 4.1, these resuspension rate models can also be used in connection with other assessment programs, such as RESRAD, that do not implement the models. When this approach is taken, the resuspension model is programmed as part of the front-end script program, which calculates the resuspension rate and passes the information to RESRAD (or any other program with which a front end is used) through an input file. The EPA models will be compared with other resuspension approaches in the work for Task 5 (Independent Calculation) and a recommendation will be made. Our present reference to the variety of approaches is not intended to make the selection prematurely, but rather to stress the point that the available programs, as they stand, are merely tools. Whichever tool is chosen must be coupled with judgment, research, and due consideration of site-specific characteristics to produce a persuasive assessment.

The resuspension pathway affects several components of radiation dose: (1) inhalation, (2) external gamma dose from airborne particles, and (3) deposition onto foliar surfaces of food and fodder crops, thus affecting the ingestion dose from consumption of local produce and animal crops. For oxides of plutonium in the soil and a scenario such as the resident rancher or hypothetical future resident, that is located in the field east of the 903 Pad, the resuspension-inhalation exposure mode is likely to be the dominant component of annual dose. Therefore, it is much more important to formulate credible approaches to modeling the resuspension mechanism and quantifying its uncertainty for the Rocky Flats site than it is to devote too much time and attention to debating relative merits of one computer tool over another.

3.1.4 Groundwater and surface water transport

In calculating the proposed soil action levels (DOE/EPA/CDPHE 1996), the groundwater and surface water pathways were dismissed because (1) surface water features (Woman and Walnut Creeks) on the site 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. In addition, the aquatic food pathway was eliminated because the streams are not capable of sustaining a viable fish population. In this section, we will discuss these assumptions and the rationale behind them, and we will examine the ramifications of dismissing the groundwater and surface water pathways in the assessment.

3.1.4.1. Overview of surface and groundwater hydrology at the RFETS. Groundwater and surface water hydrology is discussed in the Sitewide Hydrologic Characterization Report (DOE 1995). The following material was paraphrased from this document and a White Paper that discussed the vertical contaminant migration potential at the RFETS (DOE 1996).

Three hydrostratagraphic units have been defined for the RFETS. Listed in descending order these units are the Upper Hydrostratagraphic Unit (UHSU), the Lower Hydrostratagraphic Unit (LHSU) and the Laramie-Fox Hills Aquifer Hydrostratagraphic Unit (LAHU). The UHSU consists of all surficial geological deposits and Arapahoe Formation sandstones that are in hydrologic connection with overlying surficial deposits, and weathered Laramie Formation claystone bedrock. These geologic units contain the uppermost aquifers underlying the RFETS. The LHSU consists of all unweathered Arapahoe and Laramie Formation bedrock and strata including upper Laramie claystones and confining beds. The LAHU consists of all unweathered lower Laramie Formation sandstone and Fox Hills Sandstone strata that comprise the regional Laramie–Fox Hills aquifer system. The LAHU forms the upper confining bed and the 7000+ ft thick Pierre Shale forms the lower confining layer.

The UHSU extends from the surface to a depth of about 35–60 feet. Small, mostly unconfined aquifers are present in the UHSU within the alluvium, colluvium, and valley-fill alluvium that make up the unit. Hydraulic conductivity in these units span 5 orders of magnitude. The geometric mean value for the Rocky Flats alluvium, colluvium, and valley-fill are 2.06×10^{-4}, 1.15×10^{-4}, and 2.16×10^{-3} cm s^{-1} respectively. These aquifers are not considered viable for drinking water or irrigation because their well yields are quite low, typically ranging from 0.05 to 2 gallons per minute in isolated areas. Water flow is typically from west to east-northeast and follows the surface topography. Aquifers terminate where they intercept the ground surface at incised surface drainage features such as Woman and Walnut Creek and at the contact between the Rocky Flats alluvium and bedrock unconformity. Surface discharge is typically manifested in the form of a seep. There is also vertical movement downward into the LHSU.

The LHSU is composed mainly of claystone and siltstone with a few discontinuous sandstone lenses. Thickness is estimated to range between 850–870 feet. Vertical migration of infiltrating waters from the UHSU into and through the LHSU is limited by the low vertical hydraulic conductivity of this unit. Laboratory tests of core samples indicate a hydraulic conductivity ranging from 1×10^{-6} cm s^{-1} near the top of the unit to 1×10^{-7} cm s^{-1} near the bottom. Fracturing, however, can significantly increase the effective hydraulic conductivity in a relatively impermeable porous medium such as the LHSU. Fracture zones have been observed in the UHSU and LHSU and provide a viable means of moving groundwater from the UHSU to the Laramie–Fox Hills aquifer system. Faulting has also been postulated as a potential groundwater transport pathway from the UHSU and LHSU to the LAHU.

The LAHU is composed of fine to medium grained sandstone separated by a few claystone beds in the upper portion. Thickness ranges from 200 to 220 feet for the "A" and "B" sandstone that comprise the lower interval of the Laramie formation, and 80 feet for underlying Fox Hills sandstone unit. The Laramie–Fox Hills aquifer system is the target of most water wells in the vicinity of Rocky Flats because this aquifer provides sufficient water for domestic and industrial

uses. Recharge to the aquifer takes place along the foothills west of the RFETS where the permeable sandstone beds of the formation are folded up and exposed. The permeable sandstone generally dips eastward toward the center of the Denver Basin.

Surface water features at the RFETS include Walnut and Woman Creeks and several ditches that provide irrigation water. Walnut and Woman Creeks are perennial and generally respond to seasonal fluctuations in precipitation, recharge, groundwater storage, and stream and ditch flow.. In the past these creeks drained into and Standley Lake, respectively. As of 1992, Walnut Creek, which previously flowed into the Great Western Reservoir, was diverted around Great Western Reservoir. By 1996, Woman Creek no longer flowed from the site directly into Standley Lake.

3.1.4.2. Implications of ground and surface water pathways on soil action levels. In an analysis of the vertical contaminant migration potential at RFETS (DOE 1996) it was concluded that the upper Laramie Formation confining beds have a sufficient amount of hydrologic and geochemical integrity to provide long-term protection of the Laramie–Fox Hills Aquifer from contamination at the RFETS. After reviewing this document and its supporting calculations, we agree with their conclusion but do not see this as a reason to discontinue research in this area or to dismiss entirely groundwater issues at the RFETS. The analysis leaves open other potential water transport pathways, and the possibility of colloidal transport may be important. Most notably, these potential pathways include lateral transport in the UHSU and discharge to surface water features followed by migration to downstream reservoirs. Additionally, direct usage of the UHSU aquifers could also be considered. One may also argue that under an exposure scenario that assumes subsistence conditions, a water well that produces 2 gallons per minute (such as has been observed in the UHSU) would be adequate to provide drinking water and perhaps water for a few head of livestock and some limited irrigation. Failure to address these pathways quantitatively leaves open the question of their potential importance.

It is well beyond the scope of this project to address the groundwater pathway in any substantial way other than through a simple screening exercise. Sophisticated groundwater modeling is difficult and time consuming, requiring substantial quantities of field data to characterize subsurface hydrologic units. We examine a conservative calculation in order to address the question of whether or not the pathway can be ruled out of the current analysis. We activate the groundwater pathway model in the RESRAD simulations, using the site conceptual model and parameter values developed and documented in the proposed soil action level document (DOE/EPA/CDH 1996). The RESRAD conceptual site model assumes that a scenario subject uses groundwater derived from the UHSU for drinking water and some irrigation. The default RESRAD water ingestion rate of 510 liters per year was used in the analysis. Parameter values used in the assessment were reviewed and appear to be reasonable based on the information provided in the hydrogeologic characterization reports (DOE 1995).

Results for Tier 1 Action Level (85 mrem) residential exposure scenario are shown in Table 3.1.5-1. Note that action levels changed only for ^{241}Am, ^{241}Pu, and ^{234}U. In the case of ^{241}Pu, the ingrowth and ingestion of ^{241}Am is what caused groundwater ingestion doses to outweigh doses from external sources and inhalation. In the case of ^{234}U, ingestion doses are substantially higher than doses from external radiation. Dose from external radiation made up most of the total dose for ^{235}U and ^{238}U, and therefore groundwater ingestion doses had little impact. In the case of ^{241}Am, ingestion doses are substantially higher than inhalation or external doses. The highest doses for radionuclides where inclusion of the groundwater pathway made a difference (^{241}Am, ^{241}Pu, and ^{234}U) occurred 202, 222, and 379 years from the start of the simulation respectively. Highest doses

when the groundwater pathway was ignored occurred at year 0 except for ^{241}Pu, which occurred 15 years from year 0. For the radionuclides whose action levels changed when the groundwater pathway was included, the differences in the times of maximum dose reflect the transit time from the source to the aquifer. For the radionuclide given the most attention (^{239}Pu), the soil action level remained unchanged.

Table 3.1.5-1 Soil Action Levels for the Residential Exposure Scenario at the 85 mrem Level Including and not Including the Groundwater Pathway

Radionuclide	Soil Action Level without Groundwater Pathway (pCi g^{-1})[a]	Soil Action Level with Groundwater Pathway (pCi g^{-1})
^{241}Am	215	110
^{238}Pu	1529	unchanged
^{239}Pu	1429	unchanged
^{240}Pu	1432	unchanged
^{241}Pu	19830	3370
^{242}Pu	1506	unchanged
^{234}U	1738	660
^{235}U	135	unchanged
^{238}U	586	unchanged

[a] Source: DOE 1996a

The results of this exercise suggest that the rationale for dismissing groundwater as a viable pathway should perhaps be investigated further. The ongoing activities of the Actinide Migration Panel and other studies involving plutonium mobility should shed additional light on this subject. However, the results of these studies will not be available in time for completion of this work. For the purpose of calculating soil action levels, we will include the groundwater ingestion pathway for at least one of the scenarios using a model with a level of complexity similar to the one implemented in RESRAD. A more detailed evaluation is not possible with the time and budget constraints of this project. We use the principle that by protecting scenario subjects who live and use water onsite, we are protecting all other potential users because transport of activity away from the site will result in lower exposure concentrations because of dilution and dispersion.

As shown by the preceding example, the inclusion of the groundwater pathway had little impact on the overall soil action levels except for the radionuclides noted, and we expect that this will be true in future simulations because inhalation and external doses tend to outweigh ingestion doses for most nuclides. We should caution that the results this assessment of groundwater are subject to reinterpretation based on any new findings from actinide migration studies and additional investigations performed for site remediation purposes.

3.2 Exposure Modes

The exposure modes described in this section have already been mentioned in previous sections to illustrate exposure pathways. The basic modes are inhalation and ingestion (internal exposure) and exposure to an external medium containing beta- and (primarily) gamma-emitting

radionuclides. Other possible modes for internal exposure are absorption of a radioactive compound through intact skin or introduction of radioactivity into blood or by contact of a radioactive chemical with an open injury.

All types of radiation from radionuclides are significant for internal exposure. For external exposure, the dominant radiation type of a radionuclide permits some generalizations. Alpha-emitting radionuclides are not ordinarily a significant external source. Some beta emitters in high enough concentration in close proximity to a subject for a sufficient time can produce short-term damage to the skin, but beta rays have limited penetration in tissue and their dose is usually confined to a layer within a few millimeters of the skin surface. Gamma emitters produce penetrating rays that are capable of delivering energy (dose) from an external source to all parts of the body. The magnitude of the gamma dose received depends on the concentration of the gamma-emitting radionuclide in the source medium, its energy spectrum (higher energy photons tend to distribute their energy more deeply in tissue than lower energy photons), the geometry of the medium, the duration of the exposure, and the distance of the subject from the source medium.

Practical dose estimation is accomplished by means of dosimetric databases, consisting mainly of *dose coefficients* (sometimes called *dose conversion factors*) and other factors that relate the various kinds of exposures to the dose received per becquerel (Bq) of a radionuclide taken into the body or the dose rate per unit concentration of a radionuclide in an environmental medium to which a subject is exposed. These dosimetric factors are computed by specialists, who use models of physical and biological processes to simulate the interaction of radiation with tissue and the dynamics of metabolism of radioelements and compounds by organs of the body. Dose may be estimated by multiplying an intake rate (such as the breathing rate of someone inhaling a radionuclide suspended in the air, or the daily amount of a radionuclide that is being consumed with water and food) by the appropriate dose coefficient (intake per day times effective dose per unit intake = committed dose per day) and by the duration of the exposure; or by multiplying the concentration of a radionuclide in an exposure medium (such as the air) by a dose factor that gives dose rate per unit concentration of the radionuclide in air (= dose received per day) and by the duration of exposure. There is a difference of interpretation between the internal and external dose estimates just indicated by example. When a radioactive chemical is taken into the body, time is required for the chemical to be translocated to the internal organs, metabolized, and excreted. During this process, the organs and tissues are exposed to the radionuclide and receive dose, but the amount of dose depends in part on the time required for metabolic processes and radioactive decay to remove the material from the body. For some radionuclides, the time over which the dose from a single intake accumulates is measured in years, and accordingly, we speak of the *committed* dose that will result from the intake (although some radionuclides have short half-lives and are quickly removed by radioactive decay, and some radioelements and compounds have biochemical properties that cause them to be rapidly removed from the body). External dose, on the other hand, is delivered at a practically instantaneous rate as long as the subject is exposed to the medium in which the radionuclide (or other source) is distributed.

Dose can be estimated for any organ that absorbs energy from ionizing radiation. The *effective dose* is a concept promoted by the International Commission on Radiological Protection (ICRP), which gives a nonlocalized definition of dose that is roughly proportional to the risk of radiation-induced cancer in *some* organ or tissue; the proportionality is achieved by weighting the equivalent dose to each internal organ with a relative risk coefficient for the organ (ICRP 1977). The effective

dose is not to be confused with whole-body dose, which lacks this more refined connection to cancer risk.

All radiological assessment computer programs that we consider have databases of internal dose coefficients and external dose rate factors for each of a large library of radionuclides, including the relevant plutonium and americium isotopes for the Rocky Flats site and the decay products. The databases are similar among the programs, to the extent that they are based on published guidance from the International Commission on Radiological Protection (ICRP), particularly for internal dosimetry. The tables of internal dose coefficients provide alternative sets of numbers for different element-specific solubilities for both inhalation and ingestion. External dose rate factors are taken from Federal Guidance Reports such as Eckerman and Ryman (1993).

4. CANDIDATE COMPUTER PROGRAMS

4.1 Introduction

We originally selected for review five candidate computer programs that were developed for environmental risk assessment. The criteria for selection included the following:

(1) Presumed correctness of the models implemented by the programs, as indicated by their general acceptance, logical correspondence with features of the site, treatment of exposure pathways, and consistency with the available site data

(2) Amount and quality of validation that has been carried out and documented, and suitability for validation with local data

(3) Quality of program documentation and availability of source code

(4) Platform (i.e., computer and operating system) and (if source code is made available) programming language

(5) Flexibility of operating features, particularly the possibility of bypassing the user interface in order to invoke the computational part of the program and specify input and output files from the command line.

We confined the selection to programs that are generally comparable to RESRAD and that are (or are likely to be) widely used. In accordance with the contract, we include RESRAD as one of the candidates (it would have been included in any case). The other programs are MEPAS, GENII, MMSOILS, and DandD. All five have been (or are being) developed under sponsorship of one or more federal agencies, and to the best of our knowledge, the development project for each program has been carried out under formal quality assurance (QA) protocols.

The five criteria listed above were formulated before we made final decisions about the selection and before we began to procure code and documentation, install the executables on computers, and explore ways in which each program could be used. We have been allowed to see the source code for RESRAD. Source code is distributed with MMSOILS and GENII. We were not granted access to source code for MEPAS, but some version of DandD source code may be available, though it was not yet available to us as this report was prepared. It is not and was never our intention to carry out detailed reviews at source code level. We were primarily concerned with ways of executing the programs as indicated in item (5). We felt the need to be able to use scripting programs to manage Monte Carlo selection of parameter sets, to permit initialization calculations of relative abundances of plutonium and americium isotopes, and to invoke each of the five programs from the command line through the scripting program, passing each parameter selection prior to execution. This mode of operation permits us to apply Monte Carlo methods to programs that have no internal provision for them. Even with RESRAD, which has a beta-test version of a Monte Carlo facility, the built-in version is not entirely satisfactory for our purposes. RESRAD, MMSOILS and GENII are adaptable to this approach.

All five of the programs can be installed and executed under some version of the Microsoft Windows operating system (95 or NT, and presumably 98; by compiling the FORTRAN source code, we have executed MMSOILS under the Linux operating system, which is a variant of Unix; the instructions downloaded with MMSOILS indicate the installation procedure for DOS or Windows). Thus all of the programs would be widely accessible.

Comparative studies of three of these programs (RESRAD, MMSOILS, and MEPAS) have been made by groups including members who participated in their development (Laniak et al. 1997; Mills et al. 1997).

As this Task 2 report was nearing completion, a relevant report by the National Council on Radiation Protection and Measurements was released (NCRP 1999). NCRP Report No. 129 extends the NCRP series on screening limits, and this latest installment directly addresses radiation doses from exposure to contaminated surface soils. The report hypothesizes eight exposure scenarios and provides extensive tables of parameter values, screening limits, and dose estimates, with estimated uncertainties. The timing of the release of NCRP Report No. 129 did not permit us to prepare any substantial commentary on its relationship to this project. The reader should bear in mind that NCRP Report No. 129 is about screening limits. These limits are based on an annual effective dose limit of 25 mrem for exposure to a particular site, and this limit refers to the maximum dose to any exposed individual within a period of 1000 years. The screening limits (units Bq kg^{-1}) correspond to soil action levels for the NCRP-defined exposure scenarios, although the "action" envisioned in the screening context would likely consist of some level of site-specific reassessment. As we move forward with the project, we will continue to evaluate NCRP Report No. 129 for any implications that its methods and data might have.

This project's Request for Proposals (RFP) expressed concern for validation of the programs to be considered. We feel that it is necessary to go into some detail about procedures usually (but not always) termed *validation* and *verification* as applied to models and computer programs. We wish to be as clear as we can about what can and cannot be assumed with regard to procedures that are labeled with these terms.

4.1.1 Verification of Computer Programs

We believe it is necessary to make a distinction between the terms *validation* and *verification* (and the corresponding verbs) when they are applied to computer software. We need to go into some detail about these concepts, because one term is frequently used in place of the other, and usage is not uniform. Validation enters prominently into the project contract, and we need to strive for a clear understanding of what is possible in this regard and what is not.

Verification refers to procedures that try to ensure that a program is correctly coded, which is to say that it faithfully implements the mathematical descriptions of the models that define it, that it correctly translates input information furnished by the operator into all parameter values and control information required for calculations, that it detects inadmissible entries in the input, and (given admissible input) that it produces output that is in correct correspondence with the input. A process of verification would be perfect if one could somehow prove that for any set of admissible input data, the program will provide the output that the mathematical models and the algorithms imply, and that any inadmissible input data will be flagged. Computer scientists study verifiability as an academic subject and endeavor to develop methods for proving that a given program does what it is intended to do. As a practical matter, verification is an empirical process of systematic testing at many levels during development, investigating apparent anomalies reported by users, and making corrections as required. A reality that must be accepted is that all complex software is imperfect to some degree; in the vernacular of the trade, it has "bugs." The amount and quality of testing that a programming project can afford depends on the intended use of the software and the seriousness of the probable consequences, should it malfunction. When failure may cause injury, loss of life, property damage, or misallocation of significant sums of money, then extensive testing is necessary, and its cost must be supported. Different levels of criticality are formalized in QA procedures for software. The length of time a computer code has been used is perhaps a more important factor. Codes with a long track record of performance have had many of their bugs

pointed out by users and corrected by the developers. Users have also compared code output to their own hand calculations or results from other codes that perform comparable calculations. Taking this longevity into account, a user may gain confidence that the code is performing in a satisfactory way.

4.1.2 Validation of Computer Programs

Validation is an entirely different concept from verification. Validation also entails testing, although it is testing of a different kind. We will point out here that validation also has a special meaning in the realm of computer code quality assurance (QA). In this context, validation of a program is the process by with all of its modules are tested together, as a whole. The test is satisfactory if the requirements identified in the software specification and requirements documents are met. The present discussion does not address this narrower meaning of computer code validation. Instead, we consider model validation — that is, the collective ability of the mathematical models encoded in the computer program to predict the behavior of contaminants in the environment.

Abstractly, a computer program is considered valid for a specified predictive application if its results can be shown always to approximate acceptably their real-world counterparts. Thus, if we know how much uranium was released from a nuclear facility during a particular period and we have air monitoring data for uranium for that period, then using the known releases and an atmospheric diffusion model, we can predict air concentrations at the locations of the monitoring stations and compare the predicted concentrations with the measured values (if we assume that no other source of airborne uranium is distorting the measurements). If the approximation is acceptable, we have validation of the model for the period and the monitoring locations. Like verification, validation is necessarily imperfect (indeed, in a strict sense, it is impossible; *invalidation* would be decisive if the predictions and observations did not agree, but a claim of *validation* is merely a finding of no contradictory evidence, which leaves open the question of whether such evidence still might exist). The testing is specific rather than general: it is useless to declare that a computer program "has been validated," without specifying the particular comparisons that have been carried out. In our experience, validation of software that is applied to environmental assessments needs to be site-specific, and conclusions of any comparison must be drawn very cautiously. In the uranium example just mentioned, we might be willing to extend our tentative confidence in the model to other locations within the assessment domain that are not much farther from the facility than the monitoring stations, and we might accept predictions for other periods when we have data on releases but no monitoring data. But if we used the model to predict deposition of uranium on the ground near the facility without having measurements of uranium concentrations in the soil, for example, we would probably be going beyond the validation exercise that we have described, and although deposition rates are proportional to air concentrations, the predicted deposition rates would not gain the same credibility from the exercise as the predicted air concentrations.

The interpretation of validation exercises is never entirely clean. Consider once again the example of predicting uranium concentrations in air. Our calculations involve more than the computer program: there are the estimates of the uranium releases, which are subject to error, and there are meteorological data, which may or may not be accurate for the locations and period for which they were applied. It is possible for errors in the data to compensate for errors in the model, giving apparently good results and encouraging us to trust a program that intrinsically might not be

an acceptable representation of the processes we are simulating. Alternatively, errors in the data could make an acceptable model look bad. When we must depend on data that are available, it is practically impossible to implement rigorous designs that might remove these confounding effects. We must generally be satisfied with making as many tests of two or more correlated functionalities (e.g., diffusion and deposition, if we have data for both) as possible, in the hope that good agreement of predictions and data will be persuasive at an admittedly subjective level.

There are processes for which validation would require measurements spanning impractically (or impossibly) long time intervals. The rate of removal of plutonium from surface soil is a relevant example for which many years of data — possibly a century or more — at the same set of locations would be required for validating some relevant parameters of RESRAD for Rocky Flats, when the intent is to use scenarios spanning 1000 years.

The computer programs themselves sometimes thwart validation efforts. When the computed results must be interpreted as spatial or temporal averages, and the only data available for comparison are specific to a small part of the assessment domain, or represent only a brief period, then the comparisons may be meaningless. There are instances when the program does not output those quantities that would be used for comparison; this is often the case when the desired endpoint is dose or risk, but for validation, we may need predicted concentrations of radionuclides in air, soil, or water.

We do not wish to convey the impression that we believe the kinds of comparisons usually called *validation* are not important. On the contrary, we include them whenever we believe they can contribute to the level of confidence we and others might have in the application of a computer program that we are using. But we stress the point that in no circumstances should any computer program be considered "validated" in the abstract so that its output is implicitly trusted. In our view, validation is a process involving a specific problem (e.g., an environmental assessment involving specified scenarios and pathways at a particular site), analysts, other interested parties, a computer program, and sets of data that can be interpreted as exogenous inputs, parameter values, and outcomes of processes simulated by the computer program. When the people involved can agree that persuasive correlations of predictions and data have occurred, then we may consider the program to be validated with respect to the processes, data, and other specifics (e.g., location and time) that have been tested, but always bearing in mind that our sense of caution should increase as we apply the program to conditions different from those of the tests. A decisively negative result of a validation process is also a useful result (although often considered an inconvenient one), in that it points to something that is wrong about the program, the data, or the interpretations that have been made; but such a result usually produces further analysis and eventually another set of tests. And we must add that in some cases, a satisfactory validation (by which we mean that it reaches an accepted result, affirmative or negative) may not be possible.

Given the inherent difficulties of validation, one often has to supplement it with other approaches. Uncertainty analysis, appropriately applied, leads to results that quantify possible errors that derive from lack of knowledge or variability of parameters. Uncertainties about the proper structure of the model are more difficult. The temptation is to try to broaden the "space" of models from which the one in question has been drawn and to extend the uncertainty calculation to a representative set of possible replacements from this space of models (Draper 1995). But this approach has immense conceptual and technical difficulties. A more pragmatic option is to accept model structures that have been affirmatively validated in a variety of similar problems as provisionally correct but with magnitudes of uncertainty indicated by a broad range of experience.

For example, in atmospheric diffusion calculations, the straight-line Gaussian plume model is widely used in environmental applications, although this model is based on assumptions that are technically too simple for most of those applications. But experience and experiment indicate that for particular categories of predictive use, the Gaussian plume can be associated with corresponding uncertainty distributions. For example, from a review of numerous sets of experimental data, Miller and Hively (1987) concluded that for flat terrain, away from coastal areas, the Gaussian plume can predict annual averages of concentrations within a factor of two 90% of the time out to a distance of 10 km and within a factor of four with 90% probability somewhat beyond that distance. Such information must be applied with care and skill, but it provides an empirical representation of atmospheric diffusion and some level of confidence in the model; the cost is the stated uncertainty. This illustration, however, should not be interpreted to mean that the straight-line Gaussian plume model is applicable with knowable uncertainty to any atmospheric diffusion problem. It is not, and we know of no model that is.

Some scientists object to the use of the terms *verification* and *validation* (which are sometimes used interchangeably in the sense in which we have used the latter) in connection with numerical models of complicated and incompletely understood open systems (i.e., depending on incompletely specified initial and boundary conditions and exogenous information). Oreskes et al. (1994) criticize definitions given by DOE and the International Atomic Energy Agency (IAEA) in which validation implies that a model or program correctly represents a physical system, and these authors correctly emphasize that such a claim "is not even a theoretical possibility." They would prefer the use of more neutral language, replacing *verification* and *validation* with terms that indicate judgment and contextual interpretation of model performance.

4.2 RESRAD

The U.S. Department of Energy (DOE) and Argonne National Laboratory (ANL) have developed the computer program RESRAD (RESidual RADioactivity) for the purpose of performing calculations related to meeting the Department's criteria for residual radioactivity. The program originally (1989) implemented site-specific guidelines (called soil action levels in this report) based on a dose assessment methodology consistent with DOE Order 5400.5 (DOE 1993).

The most recent version of RESRAD for which we received executable code from ANL (Version 5.82, transmitted to us in October 1998) differs in some important respects from older versions that are still in use; in particular, it differs from the version of RESRAD that was used in the preparation of the action levels document (DOE/EPA/CDPHE 1996). Thus RESRAD is not uniquely defined for this study, and we must distinguish among versions of the program in discussing it and in considering it for possible use. In Sections 4.4.3 and 4.6.3, comparisons of GENII and RESRAD, and DandD and RESRAD, respectively, were made using Version 5.61 of RESRAD.

4.2.1 RESRAD overview

The manual for Version 5.0 (Yu et al. 1993), which was distributed with Version 5.82, does not correspond to the more recent graphic user interface (GUI) implementation. A user's guide for the latter, which is a replacement for Chapter 4 in the manual (Yu et al. 1993) is now available from ANL or from the web site http://www.ead.anl.gov/resrad. DOE has directed ANL to discontinue distribution of RESRAD versions for the DOS operating system, the most recent of which was

Version 5.62. Some of the information we received seemed to suggest that there might be incompatibilities of DOS versions with contemporary Windows operating systems. However, we have tested Version 5.61 in a command window under Windows NT and encountered no problems with it. However, a major algorithmic change affecting the Windows versions of RESRAD (beginning with Version 5.75) has been made in the area factor for the resuspension of soil particles (Chang et al. 1998). The difference in predicted doses and soil action levels can be significant. We will discuss the change in a later section.

The manual for RESRAD (Yu et al. 1993 with replacement for Chapter 4) is written with reasonable clarity and is a good compromise between encyclopedic detail (which nevertheless would sometimes prove helpful) and readability. Five chapters (and a sixth of references) provide introductory material, a rather good discussion of the pathway analysis implemented by RESRAD, a definition and discussion of *guidelines* for radionuclides in soil (the RESRAD and DOE term for what this report has called soil action levels), a user's guide for the program keyed to the earlier version 5.0 (for which the previously mentioned replacement is available), and a discussion of the "As Low as Reasonably Achievable" (ALARA) process. A set of appendices provides detailed information on the models and approaches incorporated into RESRAD (some of the information in Appendix B is made obsolete by the presentation of Chang et al. (1998)). A substantial index should be high on the list of priorities for this manual, and we would recommend breaking the user's guide (Chapter 4) into a separate document, which can more easily be kept current with new releases (a replacement for this chapter has been issued for the Windows versions of RESRAD).

The basic model that RESRAD implements is the family farm or homestead with soil and possibly surface water and groundwater contaminated with residual radionuclides. However, pathways (inhalation, external gamma radiation from soil and airborne radioactivity, soil ingestion, drinking water, ingestion of vegetables, meat, and milk) can be individually switched on or off to permit the treatment of other scenarios. RESRAD begins with an assumed initial mixture of radionuclides in an unsaturated soil compartment called the contaminated zone (CZ), which is a slab of finite area that may or may not be isolated from the surface by a cover layer (for applications at the Rocky Flats site, the contaminated zone has no cover layer; it is assumed to extend from the surface to a depth of 15 cm). In general, the contaminated zone is a proper subregion of the unsaturated zone. The unsaturated zone may be partitioned into as many as five independently parameterized strata to simulate soil zones with different transport characteristics, and the contaminated zone may be contained in one of these layers or intersect two or more of them. Initial radionuclide concentrations of radionuclides in the saturated zone (groundwater) may also be included. RESRAD simulates the removal of radioactivity from the contaminated zone by leaching, moving it vertically into groundwater, and by runoff into streams or ponds. If the water pathway is activated, contamination of drinking water at a central or peripheral well site is estimated, and contaminated groundwater may be mixed with contaminated surface water for drinking, household use, irrigation, and watering livestock.

Radioactivity from the contaminated zone may be resuspended by a mass-loading model; separate resuspension pathways are implemented for inhalation exposure and for foliar deposition on crops and animal fodder. External doses from exposure to gamma emissions from the contaminated zone and the resuspended contaminated soil particles are estimated. Beginning with Version 5.60, the external radiation field calculations incorporated refinements for the finite area and volume (with possibly irregular shape) of the contaminated zone, in contrast to previous

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methods that assumed semi-infinite distributions of radioactivity in source media (Kamboj et al. 1998).

As we have pointed out in Section 3.1.3, resuspension of contaminated soil at Rocky Flats should not be treated as a routine matter, and there are several approaches that need to be considered. As noted above, versions of RESRAD beginning with 5.75 represent the area factor for resuspension in a more elaborate way that potentially produces dose and soil action level estimates that differ significantly from those of earlier versions. RESRAD does not include a conventional atmospheric transport model for estimating remote air concentrations and foliar deposition (e.g., at locations away from the contaminated zone on the Rocky Flats site), but the manual gives some guidance for carrying out auxiliary calculations if they are required. However, the new approach to the area factor for resuspension (Chang et al. 1998) does make use of the Gaussian plume model, but the use of this model is confined to estimation of the area factor and thus effectively applies the Gaussian plume model only to a receptor at the downwind boundary of the contaminated zone.

Ingestion pathways for crops, meat, milk, and direct ingestion of soil are included in RESRAD, with the assumption that the food for people and fodder for animals are grown in the soil of the contaminated zone. Thus these plants are subject to radionuclide uptake through the roots and surface contamination by foliar deposition by resuspended contaminated soil. The dose conversion factors that are applied to the ingestion pathways correspond, by default, to the most readily absorbed (i.e., most soluble) form of each radionuclide that is available in the database. This means that the largest available value of the gut absorption parameter f_1 is used. For isotopes of plutonium, the RESRAD default assumption is $f_1 = 10^{-3}$, which means that approximately 1/1000 of the plutonium activity that passes through the small intestine is absorbed into body fluids and translocated to systemic organs, principally bone. Less soluble forms of plutonium, such as oxides, would correspond to $f_1 = 10^{-5}$. The analyst can decline the RESRAD default and opt for a dose conversion factor with a smaller value of f_1 from the database (provided one is available; 10^{-5} is available for plutonium). For material incorporated into plant tissue by root uptake, an argument may be made that the process favors an ionic state of the nuclide, but for oxides of plutonium that deposit on plant surfaces, $f_1 = 10^{-5}$ is likely the more realistic choice. However, the assumption of the more soluble form is a common one for screening calculations.

Area factors for crops, meat and milk account for fractions of the quantities consumed that come from inside the contaminated area, as opposed to the remainder, which is assumed to be produced elsewhere and uncontaminated. The default assumption is that at most half of the produce consumed is raised within the contaminated area; for meat and milk the fraction increases linearly to 1.0 as the area of the contaminated zone increases to 20,000 m^2. The analyst can change these default values.

Foliar deposition and retention is based on a simple steady-state model. The deposition rate is computed as the air concentration of radioactivity and a deposition velocity that depends on the assumed physico-chemical state of the material (0 m s^{-1} for relatively inert gases, 10^{-2} m s^{-1} for halogens, and 10^{-3} m s^{-1} for everything else; these values appear to be hardwired into the program). An interception fraction determines how much of the deposition flux is retained on the plant (this value may be changed), and the amount is decreased over the holdup time according to a first-order weathering rate parameter with a default value that corresponds to a half-time of about 2 weeks. The model also depends on the crop yield for the type of food (produce, fodder for meat, or fodder for milk). The air concentration on which this pathway depends is based on a mass loading model

that is similar to but evaluated separately from the one for inhalation, because the effective air concentration for inhalation depends on times spent indoors and outdoors.

RESRAD has in common with the other computer programs considered in this report — except MMSOILS — the capability of performing its calculations for radionuclides that belong to possibly long and complex decay chains. This capability involves solving generalizations of the well-known Bateman equations of decay and formation of radioactive progeny, combined with first-order removal of radionuclides and decay products from environmental compartments. Although mathematically routine, the computational details are quite tedious and susceptible to errors from loss of significant digits if the strategy is not carefully managed. For the radionuclides present in the Rocky Flats soils, the decay chains are non-trivial and make ad hoc calculations tedious.

RESRAD also provides virtually exhaustive output, summarizing all input data and database numbers and providing nearly every breakdown of output by pathways, radionuclides, dose, and concentration in media that might be desired.

4.2.2 Code acquisition

Argonne National Laboratory sent us Version 5.82 of RESRAD for Windows October 13, 1998, together with the manual for Version 5.0, with no notification of availability of updated documentation. Our request for the DOS version was declined, in a letter stating that the DOS version was no longer distributed. On October 23, 1998, the Rocky Flats Citizen Advisory Board received the computational part of the source code for Version 5.62, accompanied by a letter to Mr. Tom Marshall, Chairman, from W. Alexander Williams of the DOE Office of Eastern Area Programs, Office of Environmental Restoration, Germantown, MD. In the letter, Dr. Williams states that the computational code for Versions 5.61 and 5.62 is identical. He cautions that Versions 5.61 and 5.62 were written for the DOS operating system and are no longer distributed. Windows versions of RESRAD 5.61 and 5.62, he states, "were available for test and evaluation, [but] these versions may not be compatible with newer releases of the WINDOWS operating system." He alludes to "changes made in RESRAD to accommodate the changing computer platforms." Although the letter emphasizes changes that relate to the compatibility of RESRAD with different versions of the Windows operating system (presumably Windows 3.1 vs. Windows 95/98/NT), it makes no mention of the algorithmic differences between versions 5.62 and later versions beginning with 5.75. As we pointed out in Section 4.2.1, these algorithmic differences affect the resuspension pathway, in particular, and the resulting estimates of dose and soil action levels in potentially significant ways. We were not provided with computational source code for Version 5.75 or later.

We have developed an initial front-end program that performs preliminary calculations related to contemporary levels of plutonium, americium, and their decay products in the soil east of the 903 Pad. This front-end program writes files for RESRAD to read and then initiates the execution of RESRAD. The front-end program can execute RESRAD repeatedly in Monte Carlo fashion to obtain distributions of estimated radionuclide concentrations or annual doses to exposed scenario subjects. This particular front-end program is intended for use with the contemporary (unremediated) levels of radionuclides; variant versions will be prepared that will calculate soil action levels. Such a front-end approach permits us to substitute alternative resuspension mechanisms that RESRAD does not incorporate, as discussed in Section 3.1.3. Details of the front-end programs will be given in the Task 5 report.

If the questions of algorithmic inconsistency between the RESRAD documentation and the program can be resolved satisfactorily, we believe RESRAD can be used as the primary tool for investigating the benchmark (and possibly other) scenarios of use of the Rocky Flats site and the establishment of the relationship between radionuclide levels in the soil and annual dose standards (soil action levels, in particular). Factors that weigh in favor of RESRAD are (1) its continuing support by DOE, (2) its longevity, with a corresponding base of experience and understanding of its strengths and limitations, (3) its extensive well-formatted output, and (4) its design that permits us to separate the calculating engine from its graphic user interface and control it from a front-end scripting program. RESRAD has no monopoly on these features individually, but collectively it achieves a marginal lead over GENII, the other program that was not eliminated from consideration for this project. The inconsistencies in the distributed materials for RESRAD, however, are troubling. The fact that DOE does not choose to make the source code generally available for public inspection is also a negative consideration. If the source code were made available on a web site for downloading, it is our opinion that the useful feedback from a variety of users and programmers would result in developmental improvements and user confidence that would far outweigh whatever concerns the agency might have regarding unauthorized substitutions of code in compliance calculations.

With the reservations noted previously regarding the inter-version changes in mechanical resuspension of contaminated particles, the models offered by RESRAD are generally appropriate for application to the benchmark scenarios defined by the soil action levels document (DOE/EPA/CDPHE 1996) and to others constructed for purposes of illustration or likely to be proposed as alternatives to the benchmark set. However, as with any environmental models, they should be applied with a healthy amount of skepticism.

Use of RESRAD should not exclude the use of other similar tools or ad hoc programs when their use is indicated for comparisons needed to shed light on questions of the performance of the environmental models. This choice of a tool should not be allowed to substitute a computer program for the underlying mathematical models and scenario definitions, which are paramount. As our comparison of RESRAD and GENII illustrates (Section 4.4.3), more or less equivalent calculations can be performed with a variety of programs or combinations of programs, provided the mechanisms are understood and differences of implementations are properly allowed for. On the other hand, it is entirely possible to make erroneous calculations with the tool of choice. We must stress the continuing involvement of professional people who have experience with environmental assessments, the relevant models, and the appropriate computing tools. Despite the early expectations of the regulatory agencies, it does not seem possible to package all of this knowledge, once and for all, in a canonical computer program and prescribe its parametric application to all sites and situations without further analysis.

4.2.3. Changes in the area factor for resuspension

We have previously alluded to algorithmic changes in RESRAD, beginning with Version 5.75, that affect the resuspension mechanism. Given the importance of resuspension in the Rocky Flats context, these changes are of potentially substantial significance.

Discussion of these changes and the related mechanisms is of necessity somewhat technical. The changes involve the calculation of the area factor, which affects resuspension predictions. The area factor accounts for the dilution of locally contaminated airborne dust by uncontaminated dust resuspended from outside the contaminated area. Larger (smaller) area factors correspond to larger

(smaller) predictions of airborne contamination, which would produce larger (smaller) predictions of dose by inhalation and by external exposure to airborne gamma-emitting radionuclides. Bearing these relationships in mind, some readers may prefer to refer primarily to Figure 4.2.3-1 for a sense of the extent to which the changes might reduce RESRAD predictions of air concentration.

To understand the meaning of an area factor for resuspension, we must consider a process of suspension, balanced by deposition, of uniformly contaminated soil that occurs upwind from a receptor location at which we are interested in the air concentration. If the upwind fetch is infinite, we would anticipate a larger air concentration of radioactivity at the receptor point than would occur if the contaminated region were finite (which is what we are assuming in applications of RESRAD). The strategy in RESRAD is to estimate an air concentration that would correspond to an infinite region and correct it by multiplying it by a factor that represents the ratio of concentration due to the finite area divided by the concentration due to an infinite fetch. A value equal to this ratio must, of course, be derived in a round-about way, because the numerator of the ratio is the very concentration that we are trying to calculate. It is this ratio that is called the *area factor* for resuspension.

Before Version 5.75, RESRAD used an area factor (AF) that can be derived from a simple box model of the resuspension and deposition process (see, for example, Hanna et al. (1983), Chapter 9). If $\sqrt{A}$ is taken as the linear dimension of the contaminated region in the direction of the wind, where A is the area, the ratio defined in the previous paragraph can be shown to be

$$\mathrm{AF} = \frac{\sqrt{A}}{\sqrt{A} + \mathrm{DL}} \qquad (4.2.3\text{-}1)$$

where DL is a dilution length that depends on the deposition velocity, the mean wind speed, and the mixing height (height of the atmospheric layer over which the concentration is averaged). RESRAD generically used a default value of 3 m for the dilution length, although it should be considered a highly variable parameter (3 is the geometric mean of 0.03 and 250 m, corresponding, we are told, to surface roughness and the height of the stable planetary boundary layer, respectively; see Chang et al. (1998)).

In what the developers of RESRAD consider a more refined approach, they have developed an area factor that considers vertical and crosswind diffusion as represented by a Gaussian plume model, with gravitational settling estimated by Stokes's law (using a tilted plume to account for depletion) and wet deposition using a scavenging model. These models introduce additional parameters, such as the size distribution of aerodynamic diameters (1 to 30 μm is the size range considered in studying the variability of the area factor), particle density, rainfall rate, raindrop size, wind speed, and the dispersion coefficients σ_y and σ_z as functions of atmospheric stability and distance from the source. The point source of the Gaussian plume is integrated over the finite contaminated area, while the receptor is kept fixed at the midpoint of the downwind boundary. The corresponding concentration for an infinite area is obtained by increasing the area of the square source region until the receptor concentration converges to a maximum value.

Reference values are assumed for some of the parameters, namely rainfall rate (100 cm year^{-1}), particle density (2.65 g cm^{-3}), atmospheric stability (Pasquill-Gifford class D, which typically occurs almost half of the time), and raindrop diameter (1 mm). The model is represented by a logistic regression curve, which was fitted to data generated by calculations for a grid of points in the parameter space. The function is

$$\mathrm{AF} = \frac{a}{1 + b(\sqrt{A})^c} \qquad (4.2.3\text{-}2)$$

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where A is the area of the contaminated zone and each of the parameters a, b, and c is a function of the particle diameter (μm) and wind speed (m s^{-1}). The functional correspondence for a, b, and c is shown in Table 4 of Chang et al. (1998).

Wind speed is available as an input to RESRAD, but particle aerodynamic diameter is not. The dose conversion factors for inhalation in the RESRAD database are based on activity median aerodynamic diameter 1 μm, and the RESRAD developers have chosen to fix the particle size parameter at this value for the present. Chang et al. (1998) compare the old and new area factors (Equations 4.2.3-1 and 4.2.3-2, respectively) in a series of plots in their Figure 5, for values of the particle diameter ranging from 1 μm to 30 μm. Using the plot corresponding to 1 μm and the curve for wind speed = 5 m s^{-1} (the average for the Denver area is about 4 m s^{-1}), with a contaminated area of 10^4 m^2, the old factor exceeds the new by roughly a factor of 6; for 100 m^2, the old area factor is more than 10 times the new one. Lower wind speeds correspond to lesser discrepancies, and higher wind speeds would give larger ones. Larger areas would correspond to better agreement between the two area factors. Figure 4.2.3-1 shows a comparison of the old and new area factors for particle diameter 1 μm plotted against $\sqrt{A}$ for several values of the wind speed.

In reading the documentation of Chang et al. (1998), we could not be certain that the distinction between physical and aerodynamic particle diameters was being consistently observed. In the form of Stokes's law that is quoted, the physical diameter is the correct interpretation. But if the tabulations are then based on physical particle diameters, a physical diameter of 1 μm would not correspond to an activity median aerodynamic diameter of the same numeric value, but rather to a median diameter of about $\sqrt{2.65} \approx 1.6$ (given the assumed density of the particles). The language should be clarified.

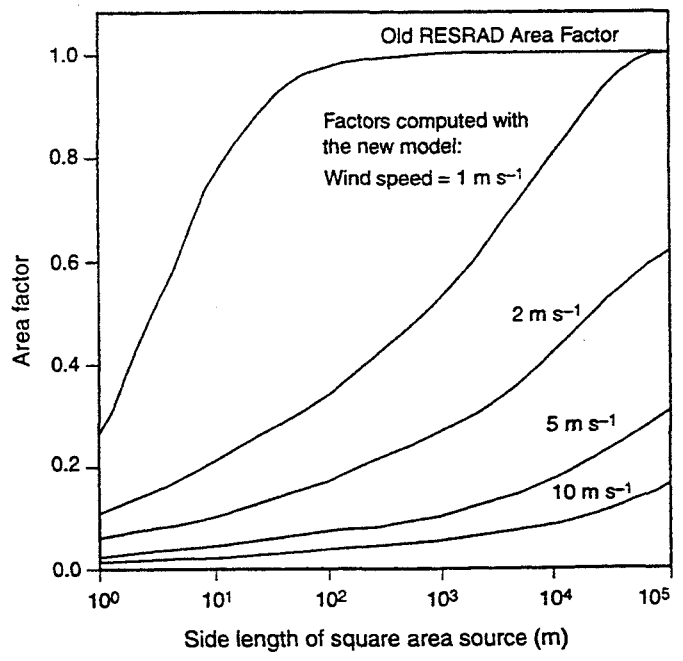

Figure 4.2.3-1. Comparison of the old and new RESRAD area factors for particle size 1 μm, plotted against the side length of a square contaminated area. The new area factor is shown for several values of the wind speed. This figure was redrawn from Chang et al. (1998).

A potentially more serious criticism concerns the generic use of this area factor in assessments at various locations with different circumstances. Perhaps in anticipation of this point, Chang et al. (1998) present a series of sensitivity calculations, varying pairs of parameters, and showing results separately for particle diameters 1, 10, and 30 μm. The variable pairs are wind speed and rainfall rate; wind speed and particle density; and wind speed and atmospheric stability. In each case, the relative area factor (perturbed divided by nominal) is plotted against the side length of the area source. The greatest variations from the nominal case occur for variations involving particle density (from 1.325 to 5.– [illegible] g cm^{-3}) and for high wind speeds in unstable air. Most variations of the relative area factor are within a factor of two, and none is as large as a factor of three.

The presentation of this sensitivity analysis may tempt a reader to the conclusion that the uncertainty introduced into resuspension-dependent quantities by the area factor is some composite of the variability shown in the figures. However, the sensitivity analysis demonstrates only the propagation of parameter variations; it does not necessarily deal with uncertainty in the models themselves relative to the real environment. For example, Miller and Hively (1987) reviewed numerous applications of the Gaussian plume model to cases where such variables as the release rate, wind speed, atmospheric stability, and downwind concentrations were monitored or could be considered known. At best, the predicted annual-average concentrations agreed with the observations to within a factor of two when the terrain was regular and the meteorology unexceptional (i.e., $0.5 \leq$ predicted / observed ≤ 2); in cases of irregular terrain or (for example) coastal meteorology, the reported annual-average uncertainty was a factor of ten. Generic application of a Gaussian plume model should involve consideration of these uncertainties. Of course, the application of the Gaussian plume to the area factor differs in scale and detail from conventional predictions of concentration downwind from a source, and in some part the uncertainty may derive from parametric uncertainties, but it seems to us that we cannot assume *a priori* that the model is intrinsically more reliable for deriving the area factor than the study of Miller and Hively (1987) has shown it to be for conventional applications.

Another point that can be raised regarding the models used to derive the area factor is that the representation of dry deposition by the Stokes's-law gravitational settling model is at best an approximation that ignores the partial dependence of the particle behavior on micrometeorological variables. For particles with aerodynamic diameter near 1 μm, Stokes's law may not be an adequate parameter for total deposition for purposes of the area factor.

It is not our intent to criticize the RESRAD developers. The models and parameters that they have applied to estimate the area factor are well known and frequently invoked. Their approach is rational from a research standpoint, their analysis seems thorough, and we are appreciative of the well-organized numerical explorations they have provided in Chang et al. (1998). Our reservations have more to do with objections to generic application of assessment models. The developers consider this formulation of the area factor more realistic than the older version that was based on a simple box model (Equation 4.2.3-1), and that may be true. But in any assessment, the analyst should be weighing the appropriateness of any factor that enters into the calculations for the site in question and integrating each factor into the composite uncertainty picture. We certainly agree with the last sentence in Chang et al. (1998): "However, if measurement data are available, the measured air concentrations [*sic*] data should be used in RESRAD analysis." The user's manual should clarify just how this is to be done; we assume it would involve supplementary off-line calculations based on RESRAD output. We will be making use of such measurements in the calculations for Task 5.

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In general, one can expect Versions 5.75 and newer of RESRAD to predict lower annual resuspension-dependent doses and correspondingly larger radionuclide soil action levels, with the extent of the discrepancy depending on the values supplied for the mean wind speed and the area of the contaminated zone. For application to the Rocky Flats site, we cannot make a more definite statement at this time, until an appropriate area for the field of contamination is determined. In regard to the version of RESRAD that will be applied, there is some ambiguity about the intentions of the regulatory agencies. The soil action level document (DOE/EPA/CDPHE 1996) presents RESRAD parameters and computed soil action levels that appear to correspond to an earlier version of the code (perhaps 5.61 or 5.62). This was probably the most recent version available at the time that document was prepared. But if the assessment were to be carried out in a purely formal manner, with the newer version of the code being substituted and executed with the same set of parameters, the foregoing analysis indicates that a possibly important change in the predictions would occur.

4.3 MEPAS

The Multimedia Environmental Pollutant Assessment System (MEPAS) was developed at Pacific Northwest Laboratory under DOE sponsorship. Offered as a commercial product by Battelle Memorial Institute under a technology-transfer agreement with DOE, MEPAS is the most ambitious of the programs considered here. It advertises applicability to both chemical and radioactive pollutants, with computation of human health risk for carcinogens and hazard quotients (sometimes called hazard indices) for noncarcinogens. MEPAS includes air transport models in addition to surface water and groundwater transport, and it treats all major exposure pathways (Buck et al. 1995). As we mentioned in Section 3.1.3, MEPAS incorporates variants of the EPA models for particulate suspension by mechanical and wind-driven erosion (Battelle Memorial Institute 1997). The MEPAS documentation that we have reviewed does not indicate an intrinsic Monte Carlo capability for uncertainty analysis.

Battelle Memorial Institute declined our request for permission to examine portions of the MEPAS source code. Absent special instructions, such access would be necessary to allow us to discover how to circumvent the graphic user interface and prepare a front-end interface program to provide Monte Carlo simulations and initial calculations. Accordingly, we cannot give further consideration to MEPAS at this time for application to the Rocky Flats site soil contamination. This decision was taken for reasons of practical necessity; it does not deny the potential applicability of the MEPAS models to the problems we are considering. However, it is not clear that MEPAS would offer any decided advantage over RESRAD or GENII for the specific calculations that we are considering. The wealth of models and options that MEPAS offers would likely be wasted, for the most part.

Considerable effort has gone into benchmarking MEPAS with RESRAD and MMSOILS (Laniak et al. 1997; Mills et al. 1997). In response to our request for source code access, we were sent the report of Cheng et al. (1995), which presumably is a more detailed account of the work reported by Laniak et al. (1997) and Mills et al. (1997), and what appears to be a prepublication copy of a report without a cover page, with the title *Test Plan and Baseline Testing Results for the MEPAS Saturated Zone (Aquifer) Transport Model*. These reports did not reach us in time to permit a proper examination of them, and we do not comment further on them at this time.

4.4 GENII

At the direction of the U.S. Department of Energy in 1988, the Hanford Environmental Dosimetry Upgrade Project was undertaken by Pacific Northwest Laboratory to incorporate the internal dosimetry models recommended by the International Commission on Radiological Protection into updated versions of the environmental pathways models used at Hanford. The resulting second generation environmental dosimetry computer codes were compiled in the Hanford Environmental Dosimetry System — Generation II or GENII (Napier et al., 1988). The GENII system was developed by means of tasks designed to provide a state-of-the-art, technically peer-reviewed, documented set of programs for calculating radiation doses from radionuclides released to the environment.

4.4.1 Code overview

The GENII system was designed to address exposure and dose resulting from both routine and accidental releases of radionuclides. Doses may be calculated on an annual, committed, or accumulated basis. Transport pathways include air, soil, biotic, surface water, and to a limited extent, drinking water. Pathways of exposure include direct or external exposure via water (swimming, boating, and fishing), soil (surface and buried sources), and air (semi-infinite and finite infinite cloud geometries), inhalation pathways, and ingestion pathways. The inhalation pathway includes direct inhalation of material released to the air from a facility or operation, and inhalation of resuspended contamination from the soil. Ingestion pathways include soil, and transfer of radioactivity from soil to food products (produce, milk, meat, and poultry), and contaminated drinking water.

GENII includes options for calculating both near-field and far-field (some refer to near-field as onsite and far-field as offsite) exposure scenarios. In a near-field scenario, the focus is on the doses an individual could receive at a particular location as a result of initial contamination or external sources at that location. A far-field scenario considers the doses received by an individual or a population exposed to radioactivity that has been released and transported from a location remote from the receptor. The two types of scenarios are not mutually exclusive, and any given scenario may have components of both the near- and far-field scenarios.

The proposed soil action levels developed for the RFETS are essentially based on a near-field scenario. The RESRAD code is not capable of addressing directly what GENII defines as a far-field scenario, and therefore, GENII appears to have an advantage as a model that may provide dose estimates to off-site individuals. Far-field scenarios in GENII include chronic and acute atmospheric releases, and chronic and acute surface water releases. Doses from ingestion of contaminated groundwater may be calculated in GENII, but groundwater concentrations must be computed externally to the code, using a model suited to that type of computation or direct measurements.

Source term input to GENII may be in the form of effluent release rates to various environmental media (air, soil, or water), or initial contamination levels in these media. The code allows for environmental transport calculations to be performed externally to GENII and the results input by way of a dispersion factor or a user-defined concentration value in an environmental medium. Radioactive decay and formation of decay products are handled within the code. Half-lives, dose conversion factors, and animal and plant uptake factors are stored for a library of 251 nuclides. In addition, the decay chain is automatically constructed once a parent nuclide is selected, and decay and formation of progeny are calculated for the entire decay chain over time.

The GENII package of codes was developed under a stringent QA plan based on the American National Standards Institute (ANSI) standard NQA-1 (ASME 1986) as implemented in the PNL Quality Assurance Manual PNL-MA-70[1]. All steps of the code development have been documented and tested. Extensive hand calculations have been performed and are available for review on request

4.4.2 Code features relevant to calculating soil action levels for Rocky Flats

GENII models the same pathways that are included in the RESRAD simulations that were used in the soil action levels document (DOE/EPA/CDPHE 1966). These pathways are resuspension and inhalation of contaminated soil, inadvertent soil ingestion, transfer of radioactivity into homegrown produce and animal products, and external exposure of the subject to surface soil contamination and contaminated airborne particles. Two resuspension models are available in GENII: a mass loading approach that is similar to the one in RESRAD Versions prior to 5.75, and a time-dependent method developed by Anspaugh et al. (1975). The Anspaugh model was calibrated to empirical data that showed a decrease in the amount of resuspended material over time. It appears that the Anspaugh model is not applicable to the Rocky Flats environs because it applies only to the first 17 years following a deposition event. In the case of the soil at Rocky Flats, the contamination has been there for more than 30 years.

External exposure in GENII is calculated using a modified version of the ISOSHIELD code (Engel et al. 1966). The ISOSHIELD code uses the commonly accepted techniques of Rockwell (1956) or other standard references for computing exposure rates from isotopes distributed in various geometric configurations. The calculation considers the initial photon, energy spectrum, material properties in the source region, air, and any shielding materials placed between the source and receptor (such as a cover layer of soil), and mass attenuation and build-up within the source and shield materials. Exposure rates (in Roentgen per hour) are converted to effective dose equivalents using the energy-dependent surface-dose to organ-dose conversion factors derived from information in Kocher (1981). Organ weighting factors were obtained from ICRP 26 (ICRP 1977).

Two models are available for ingestion of contaminated crops. These models are a chronic exposure model and an acute exposure model. The chronic exposure model assumes a constant source of contamination released to the model domain. The acute model assumes an initial contamination level in soil and water that is not replenished over time. The acute model appears to be appropriate for the Rocky Flats site, because the site will be shut down and release no additional radioactivity (other than what is currently present) to the environment. The acute model of GENII is conceptually similar to the PATHWAY model (Whicker and Kirchner 1987) but uses fewer inputs. It includes the processes of root uptake, recycling of contamination on the plant surface with the surface soil, redistribution due to tilling, and translocation of contamination from non-edible to the edible portions of the plant. GENII also includes models for calculating transfer of radioactivity from the soil to animals and animal products, such as milk meat, eggs, and poultry. These pathways were not considered in the original conceptual model defined for the proposed soil action levels, but it is conceivable that alternative scenarios might include them.

GENII also considers an on-site groundwater pathway like RESRAD. However, RESRAD computes transport from the source, through the vadose (unsaturated) zone, and into the aquifer

[1] Procedures for Quality Assurance Program, PNL-MA-70. This is a controlled document used internally at PNL. Information regarding the manual may be obtained from Pacific Northwest Laboratories, Richland, Washington.

while GENII only allows the user to input a previously measured or modeled groundwater concentration, and dose calculations are performed on that basis. In RESRAD, the groundwater model consists of relatively simple representations of subsurface aqueous flow and transport and does not consider off-site transport of contamination in the aquifer.

The internal dose conversion factors provided in GENII are calculated based on the models for dosimetry reported in ICRP Publication 30 (ICRP 1979–1982). These models for dosimetry were coded into the INTDF code to allow for dose to be calculated on an annual (as opposed to committed) basis for different commitment periods. While this is an important feature of the GENII code, the need to calculate dose at this level of detail is not necessary for meeting the dose requirements for soil action levels. The annual dose limit specified for the soil action levels includes the 1-year effective dose equivalent from external radiation sources and the 50-year committed effective dose equivalent from one year's exposure to internal (inhalation and ingestion) sources. Therefore, only the dose conversion factors representing the 50-year committed dose equivalent are needed for this calculation.

4.4.3 Code acquisition and testing

The GENII computed dose system and documentation, version 1.485 was obtained from the Radiation Safety Information Computational Center (RSICC) at Oak Ridge National Laboratory. The code was written in FORTRAN, and source code was provided in the distribution. The code was installed on a personnel computer running under Windows 95© and MS DOS© version 6. Primary input to the GENII software package is through an ASCII input file that may be prepared using a menu-driven pre-processor written in BASIC called APPRENTI. Other files containing dose conversion factors, environmental transport factors, and default parameter values are required for execution and are stored in the GENII default subdirectory. These files may be modified by the user using a standard ASCII text editor.

In order to test the code and observe its performance, we set up a GENII simulation assuming the same conceptual model that was used to define the proposed soil action levels for the resident exposure scenario at the Rocky Flats site (DOE/EPA/CDPHE 1996). These results could then be compared to the RESRAD Version 5.61 results, permitting us to highlight differences in the transport, exposure and dosimetry models used between the two codes. Key input parameters applicable to both codes are described in Table 4.4.3-1. Dose conversion factors used in GENII assumed the same lung clearance class and gut absorption fraction as in the RESRAD simulations used to develop the soil action levels reported in DOE (1996). This required several GENII simulations, because in any given GENII simulation, all radionuclides are assumed to have the same lung clearance class and gut solubility. Plant-to-soil concentration ratios were left at their respective default values for each code. Results were normalized to their dose per unit concentration in surface soil (mrem (pCi $g^{-1})^{-1}$) or their dose-to-soil ratio (*DSR*) for ease of comparison.

Table 4.4.3-1. Key Input Parameters for the Proposed SAL Conceptual Site Model[a]

Parameter	Value	Units
Area of contamination[b]	>1250	m^2
Thickness of contaminated zone	0.15	m
Density of contaminated zone	1.8	$g\ cm^{-3}$
Time of assessment (time after institutional control)	0	years
Inhalation rate	7000	$m^3\ y^{-1}$
Mass loading factor	2.65×10^{-4}	$g\ m^{-3}$
External gamma shielding factor	0.8	---
Fruits, nonleafy vegetables & grain consumption	40.1	$kg\ y^{-1}$
Leafy vegetable consumption	2.6	$kg\ y^{-1}$
Soil ingestion rate	70	$g\ y^{-1}$
Lung clearance class for americium	W	---
Lung clearance class for plutonium and uranium isotopes	Y	---
Gut absorption fraction, plutonium isotopes	1.0×10^{-5}	---
Gut absorption fraction, americium isotopes	1.0×10^{-3}	---
Gut absorption fraction, uranium isotopes	5.0×10^{-2}	---
Mass loading for foliar deposition	1.0×10^{-4}	$g\ m^{-3}$

a. from DOE (1996), Attachment I
b. Area of contamination in GENII is only defined in terms of less than or greater than 1250 m^2

The results (Tables 4.4.3-2 and 4.4.3-3) indicate that there is not much difference between the *DSRs* calculated with the two codes for the inhalation and ingestion pathways. However, significant differences were noted for the external exposure pathway and in particular, for ^{238}U and ^{241}Pu. The *DSRs* for these two nuclides were significantly smaller for the GENII simulations compared to those of RESRAD Version 5.61. It is not clear whether these differences were due to the photon transport and attenuation models employed in the codes or the methodology to convert exposure rate to effective dose equivalent. Differences as high as 12.4% were also noted in the ingestion pathway for uranium and americium isotopes. These differences may be attributed to differences in the terrestrial food chain models and perhaps to a smaller extent to the dose conversion factors used. The inhalation pathway showed the least amount of difference between the *DSRs* calculated with the two codes. The maximum difference between GENII and RESRAD *DSRs* was 2.9% for ^{242}Pu. Because both codes use virtually identical resuspension models that make use of the mass loading factor, the difference between the two results can mostly be attributed to their respective dose conversion factors. In terms of the *DSR* for all pathways of exposure (external, inhalation, and ingestion), differences >5% were noted only for the uranium isotopes. For the most part, RESRAD provided a more conservative estimate of dose, except for ^{241}Am and ^{234}U, where GENII ingestion doses were higher compared to those calculated by RESRAD. In general, inhalation was the dominant pathway; however ingestion was equally important for the uranium isotopes. According to RESRAD Version 5.61, external exposure was the most important pathway for ^{238}U.

Table 4.4.3-2. Dose-to-Soil Ratios (*DSR*, mrem (pCi g^{-1}) ^{1}) for RESRAD V. 5.61 and GENII

Radio-nuclide	RESRAD External	RESRAD Inhalation	RESRAD Ingestion	RESRAD Total	GENII Results External	GENII Results Inhalation	GENII Results Ingestion	GENII Results Total
Am-241	.0344	.0811	.282	.397	.0230	.0800	.310	.413
Pu-238	.00012	.0526	.00384	.0566	.00010	.0520	.00370	.0558
Pu-239	.00023	.0563	.00401	.0605	.00022	.0550	.00380	.0590
Pu-240	.00012	.0563	.00401	.0604	.00010	.0550	.00380	.0589
Pu-241	.00001	.00091	.00006	.00098	2×10^{-10}	.00089	.00006	.00095
Pu-242	.00010	.0536	.00381	.0575	.00008	.0520	.00360	.0557
U-234	.00032	.0241	.0249	.0493	.00030	.0240	.0280	.0523
U-235	.583	.0225	.0235	.629	.390	.0220	.0260	.438
U-238	.100	.0216	.0237	.145	.00014	.0210	.0260	.0471

Table 4.4.3-3. Percent Difference[a] Between the DSRs for RESRAD V. 5.61 and GENII

Radionuclide	External	Inhalation	Ingestion	Total
Am-241	33.10%	1.40%	−10.06%	−3.98%
Pu-238	16.67%	1.20%	3.60%	1.39%
Pu-239	3.51%	2.29%	5.20%	2.49%
Pu-240	14.38%	2.29%	5.20%	2.51%
Pu-241	100.00%	1.82%	7.20%	3.62%
Pu-242	17.32%	2.89%	5.44%	3.09%
U-234	4.76%	0.50%	−12.39%	−5.98%
U-235	33.07%	2.14%	−10.61%	30.33%
U-238	99.86%	2.64%	−9.79%	67.57%

a. |(DSR (RESRAD) – DSR (GENII)|/DSR (RESRAD)

4.5 MMSOILS

Developed for screening analysis of hazardous waste sites, MMSOILS was developed by the EPA's Office of Research and Development, National Exposure Research Laboratory, Ecosystems Research Division, Regulatory Support Branch and is currently available from EPA's web site in Version 4.0. Written in FORTRAN-77 and distributed with full source code and documentation, the MMSOILS program may be implemented under Windows or Unix operating systems. The accompanying documentation, which includes a user's guide and descriptions of the models, is detailed and extensive (EPA 1996).

The MMSOILS goal is estimation of human exposure and health risk from chemically contaminated hazardous waste sites. Collectively, the models of MMSOILS provide a multimedia tool that simulates chemical transport in the atmosphere, soil, surface water, groundwater, and the food chain. It treats inhalation of airborne volatile and particulate materials, drinking contaminated water, ingestion of soil, and consumption of crops and animal products that were produced on contaminated land. The program includes a Monte Carlo mechanism for propagating parameter

uncertainties into estimates of exposure and risk. MMSOILS has been benchmarked with RESRAD and MEPAS (Laniak et al. 1997; Mills et al. 1997).

It is possible to apply MMSOILS to radionuclides in the soil, but the program has no mechanism, beyond simple radioactive decay, for dealing with decay chains. Allowing for the possibility that we might be able to simulate this mechanism by pre- and post-processing methods, we included MMSOILS in the list of programs to be considered. But as a practical matter, given the time constraints of this project, such an approach would not be satisfactory. In these circumstances, we must rule out the use of MMSOILS for estimating dose and developing soil action levels for the Rocky Flats site.

4.6 DandD

The software package *Decontamination and Decommissioning* (DandD) was designed by the U.S. Nuclear Regulatory Commission (NRC) as a user-friendly analysis tool for NRC rulemakers and facilities under NRC regulation seeking decommissioned status. The code incorporates the information contained in NUREG/CR-5512, Volume 1, and helps NRC licensed facilities determine the level of cleanup required to allow the release of their property for unrestricted use.

4.6.1. Code overview

DandD was designed as a screening level analysis program to provide a simplified estimate of the dose to an average member of a carefully specified critical screening group (Daily 1999). The estimate is designed to be "prudently conservative" but is not designed to be used as an estimate of actual dose (NRC 1992).

The DandD code includes four exposure scenarios: building renovation, building occupancy, drinking water, and residential. For the residential scenario, the pathways included are external exposure, inhalation, drinking water ingestion, ingestion of food grown from irrigated water, land-based food ingestion, soil ingestion, and fish ingestion. The pathways are hard-wired into the scenarios and can only be removed from consideration by zeroing the annual intake of any given product.

Input parameters for each of the DandD scenarios have default values that were selected in such a way as to be "prudently conservative" (NRC 1992). The default values were chosen for a select and limited population group, and are not intended to represent the average over an entire population. DandD does allow modification of each parameter value within a limited range. Parameter values that are outside the range of allowed values are not accepted as input to the code. These ranges were selected using an analysis done by Sandia National Laboratory in 1997 and 1998. NRC warns that use of this conservative generic approach requires a great deal of professional judgment and common sense (NRC 1992). The intent of the code is to account for the majority of potential land and structural uses, and the code is designed to overestimate the most probable annual dose.

Doses calculated with DandD are total effective dose equivalent (TEDE) estimates, which include annual effective dose and committed dose equivalent during each year. The dose reported in the output of the calculation is the committed dose for the year of maximum total committed dose. This is comparable to the dose limit input in RESRAD (e.g. for the Rocky Flats calculation, 15 or 85 mrem according to the scenario being considered).

Source term input to DandD is strictly in the form of initial concentrations of radionuclides in soil. Radioactive decay and progeny ingrowth are calculated within the code. Half-lives, dose conversion factors, and organ specific dose conversion factors are not available as inputs within the code and remain fixed throughout the calculations. In keeping with the "prudently conservative" goal of the code, the chemical form of the radioactive material that would confer the largest dose is assumed to exist in all cases. For plutonium, this means that the most soluble form of plutonium is assumed, and the dose conversion factors used by DandD correspond to this form (clearance class W for inhalation and $f_1 = 10^{-3}$).

It is important to point out that DandD is in Version 1.0 and has not yet undergone extensive scrutiny or use. Documentation that accompanies the code has not been published, nor has the source code been publicly released. This makes it difficult to use the code and even more difficult to make confident statements about how the code functions. The release of this documentation is not scheduled to occur within a time that would allow consideration of DandD for use in this project. *RAC* has requested and awaits receipt of all code documentation and source code material upon its publication.

We have gone forward with our analysis of this code in a limited fashion to show some of the limitations of the code in its present form for application to this project.

4.6.2. Code features relevant to calculating soil action levels for Rocky Flats

DandD models most of the same pathways as RESRAD, but some of the details about the pathway analyses have been difficult to determine without supporting documentation.

Resuspension and inhalation of contaminated soil are modeled in DandD using a mass loading model that appears to be similar to the one in RESRAD Versions earlier than 5.75, but using an additional level of detail. DandD partitions residential scenario annual activity into three different categories that are accompanied by three different mass loading factors and three different breathing rates. The three categories are indoor, outdoor, and outdoor gardening. We do not have information about how area factors are handled.

The contamination of vegetables, fruits, and roots is represented by two mechanisms: foliar mass loading of resuspended soil and root uptake of contaminated soil. The most significant difference between the way RESRAD and DandD model contamination of food products from contaminated soil has to do with the soil to plant resuspension and deposition pathway.

DandD assumes a constant ratio between radionuclide concentrations in plants and soil, using a default mass loading value of 0.1 pCi g^{-1} dry plant per pCi g^{-1} dry soil. This parameter value means that plant foods are assumed to be 10% soil by weight, a rather high estimate. DandD further applies a translocation fraction of 1.0 for contamination deposited on leafy vegetables, which means that all of the soil deposited on the leaves is integrated into the edible portions of the plant.

The RESRAD model assumes a constant deposition rate with removal controlled by a first-order weathering constant (NRC 1998). The deposition and removal are assumed to occur over the entire growing season. For radionuclides without a high degree of root uptake, like plutonium, the mass loading factor in DandD dominates the ingestion dose and the total dose for the year of maximum dose. This factor seems to be controlling the dose from radionuclides without a high degree of root uptake and causing doses calculated with DandD to be higher than those calculated with RESRAD.

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4.6.3. Code acquisition and testing

The DandD Version 1.0 windows-based executable file was downloaded from the NRC web site. Supporting documentation has been requested from NRC but not yet received. The code was written in the FORTRAN programming language, and *RAC* expects to receive the source code upon its release for public distribution later this month. Input to the DandD code is provided by the user through a graphic user interface.

To test and observe the performance of the DandD code, we attempted to reproduce the hypothetical residential scenario used at Rocky Flats to calculate soil action levels (DOE 1996). This was somewhat difficult to do, as a result of the variant definitions of inputs between the two codes and the fact that some parameters used in the Rocky Flats analysis were outside the allowed distributions of parameter values in DandD or were treated as constants by DandD and could not be altered. The difference between the results are highlighted below, but the reasons are not always known, since the documentation has not yet been published and the models are not transparent.

Table 4.6.3-1 shows some of the key parameters used in each calculation. Since the DandD code uses Class W (soluble) plutonium for inhalation and a gut adsorption fraction for ingestion of 10^{-3}, the Rocky Flats RESRAD calculation was changed so that solubility class matched the DandD values (RESRAD Version 5.61 was used). This was the only change necessary to make in the Rocky Flats calculation. All further changes were made to the DandD input parameters.

Because it is not possible to inactivate pathways in DandD the way it is in RESRAD, a number of parameters were set to zero to simulate this. To match the DOE Rocky Flats RESRAD calculation, the parameters that control the pathways for meat, milk, poultry, and aquatic food ingestion, as well as the ground and surface water pathway, were set to zero.

Table 4.6.3-1. Key Input Parameters for the RESRAD V 6.1 to DandD Comparison

Parameter	RESRAD value	DandD value
Thickness of contaminated zone	0.15 m	0.15 m
Density of contaminated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Time of assessment (after shut down)	0	0
Inhalation rate	7000 m^3 y^{-1}	0.8 m^3 h^{-1}[a]
Mass loading factor for inhalation	2.65×10^{-5} g m^{-3}	2.65×10^{-5} g m^{-3}
Fruit, nonleafy vegetables & grain consumption	40.1 kg y^{-1}	40.1 kg y^{-1}
Leafy vegetable consumption	2.6 kg y^{-1}	2.6 kg y^{-1}
Soil ingestion rate	70 g y^{-1}	0.095 g day^{-1}[b]
Lung clearance class, americium	W	W
Lung clearance class, plutonium isotopes	W	W
Lung clearance class, uranium isotopes	Y	Y
Gut adsorption fraction, americium	1.0×10^{-3}	1.0×10^{-3}
Gut adsorption fraction, plutonium isotopes	1.0×10^{-3}	1.0×10^{-3}
Gut adsorption fraction, uranium isotopes	5.0×10^{-2}	5.0×10^{-2}

[a]DandD input units shown; this converts to the same value as the RESRAD parameter.
[b]DandD input units shown; this converts to half the RESRAD parameter, but DandD parameter distributions would not allow the RESRAD value, so the calculation was run with this input and soil ingestion dose from DandD was multiplied by 2.

An important parameter that could not be reconciled between the two codes is the mass loading for foliar deposition. As described above, the pathway for contamination of plants from resuspension of contaminated soil is quite different between the two models. In creating dose to soil concentration ratios for RESRAD and DandD for Table 4.6.3-2, the DandD code was run twice for each radionuclide using the above parameters. In the second run, the value for the foliar mass loading was reduced from the default value by a factor of 10 to display the large effect that this parameter has on the outcome of the calculation. Foliar mass loading in DandD is in units of picocuries per gram of dry plant matter per picocurie per gram of dry soil. The impact of this change on the dose to soil concentration ratio is shown in Table 4.6.3-2. Even with the factor of 10 reduction, the total dose to soil concentration ratios are still significantly higher for DandD than RESRAD. Table 4.6.3-3 shows the percent difference between the dose to soil concentration ratio for RESRAD and DandD.

Without the appropriate documentation, it is not possible for us to acquire a proper understanding of the models and parameters employed in DandD. This lack of available documentation precludes further consideration of DandD in this analysis.

Table 4.6.3-2. Dose-to-Soil Concentration Ratios (DSR, mrem (pCi g^{-1})$^{-1}$) for RESRAD and DandD

			RESRAD		
Radionuclide	External	Inhalation	Plant ingestion	Soil ingestion	Total
Am-241	.0344	.0796	.0269	.255	.396
Pu-238	.00012	.0703	.0237	.224	.318
Pu-239	.00023	.0769	.0262	.248	.351
Pu-240	.00012	.0769	.0262	.248	.351
Pu-241	.000015	.00148	.00051	.0048	.0068
Pu-242	.00010	.0737	.0249	.235	.334
U-234	.00032	.0237	.0051	.0198	.0489
U-235	.583	.0221	.0048	.0187	.628
U-238	.100	.0212	.0049	.0188	.145

			DandD			
Radionuclide	External	Inhalation	Plant ingestion (ML = 0.1)	Plant ingestion (ML = 0.01)	Soil ingestion	Total (ML = 0.01)
Am-241	.0443	.147	4.3	.445	.252	.89
Pu-238	.00015	.13	3.75	.37	.222	.73
Pu-239	.00029	.142	4.17	.419	.246	.81
Pu-240	.00029	.142	4.17	.419	.246	.81
Pu-241	.00005	.00279	.0829	.00834	.00484	.016
Pu-242	.00013	.136	3.96	.398	.232	.77
U-234	.00041	.0439	.347	.0472	.0297	.11
U-235	.748	.0407	.328	.0445	.0186	.85
U-238	.11	.0393	.329	.0446	.0185	.22

Table 4.6.3-3. Percent Difference[a] Between the DSRs for RESRAD and DandD

Radionuclide	External	Inhalation	Plant ingestion (ML=0.1)	Plant ingestion (ML=0.01)	Soil ingestion	Total (ML=0.01)
Am-241	−28.8%	−84.7%	−15800%	−1550%	1.18%	−125%
Pu-238	−26.7%	−84.9%	−15800%	−1490%	0.89%	−129%
Pu-239	−20.6%	−84.7%	−15800%	−1490%	0.81%	−131%
Pu-240	−145%	−84.7%	−15800%	−1490%	0.81%	−131%
Pu-241	−263%	−88.5%	−15800%	−1490%	−1.04%	−136%
Pu-242	−27.5%	−84.5%	−15800%	−1490%	1.28%	−131%
U-234	−28.9%	−85.2%	−6690%	−824%	0.51%	−125%
U-235	−28.3%	−84.2%	−6690%	−821%	0.54%	−35.4%
U-238	−13.0%	−84.9%	−6690%	−818%	1.59%	−51.7%

[a][DSR(RESRAD) – DSR(DandD)] / DSR(RESRAD)

5. CONCLUSIONS AND RECOMMENDATIONS

It seems clear from the tests and comparisons reported in Section 4 that either RESRAD or GENII could be adapted for purposes of the project. Because of its earlier stage of development and still limited documentation, DandD cannot be counted on in the time available for this project. In addition, the strong orientation of DandD to screening calculations would make it less suitable for the kind of assessment that is envisioned for Rocky Flats. MEPAS and MMSOILS were ruled out on other practical grounds.

RESRAD and GENII are based on similar models, for the most part, and the agreement of their results for the same scenario is not really surprising. The change in the RESRAD area factor for resuspension beginning with Version 5.75 is a complication. We have confined our comparisons to pre-5.75 versions of RESRAD. It is possible to circumvent the resuspension area factor with the earlier versions of RESRAD, thereby permitting the substitution of other resuspension models, but this may be more complicated with the new algorithm.

We want to emphasize one last time that none of these computer programs can guarantee the "right answer." It could be argued that there is no such thing. These programs are tools, which, in the hands of careful analysts, can be useful for carrying out the relevant computations for an assessment, or when used in the absence of proper analysis can produce misleading information. It now appears that either RESRAD or GENII applied with experience, skill, careful consideration of site conditions and data, and with proper interpretation and communication of the results, can help to complete a persuasive assessment of the RFETS. Analysts will have make adjustments for the differences in the two programs, but used properly, they should lead to similar results. RESRAD provides a more complete listing of database quantities in its output, and some of its defaults regarding inhalation solubility classes and gut absorption factors for the radionuclides considered in a run are more easily changed by the operator. For the assessment at hand, it seems fair to say that RESRAD is the more convenient tool, but GENII may have conceptual or operational advantages in other situations.

When RESRAD is applied to the resuspension pathway, we recommend that it be with full awareness of the effect of the area factor. As we mentioned in Section 3.1.3, measured air concentrations of some of the radionuclides in the source term are available, and careful consideration should be given to using these measurements or calibrating the model to them. This approach may require manipulating the input parameters so that the area factor is effectively 1. Similar manipulations will be required if alternative resuspension models are to be substituted. With some auxiliary calculation, it may also be possible to make RESRAD more useful for application to off-site scenarios.

We want to suggest that everyone concerned with this assessment pay less attention to soil action levels and instead concentrate on the relationship between particular measured or hypothetical sets of radionuclide concentrations in soil and the predicted maximum annual dose to each scenario subject. When uncertainties in environmental parameters are introduced, soil action levels will become more cumbersome to deal with and will offer little, if any, advantage.

We have some recommendations for DOE and the developers of RESRAD. We are aware that the evolving Windows graphic user interface (GUI) is intended to make the program more accessible to a variety of users, but this greater utility comes at a cost to some potential users. It often is desirable to link programs together, with outputs from one becoming inputs to another. The procedure is usually implemented by writing scripts, which are control programs for the process (Unix operating systems are particularly hospitable to this approach). But a GUI defeats script-

driven executions. We are not suggesting that the GUI be eliminated, because it is probably the preferred access for the majority of users, but we do urge DOE and the RESRAD developers to facilitate a way of bypassing the GUI and launching RESRAD from the command line.

The pieces for this mode of interaction are already in place. The GUI is currently implemented as a separate program, which interacts with the user and the database files and ultimately writes input files for a separate program, RESMAIN3, which the GUI executes through the operating system. RESMAIN3 is the computational engine for RESRAD and is executable from the command line. It reads two auxiliary files, which provide information needed for dynamic allocation of storage arrays, and it reads a data input file specified from the command line (the GUI writes this file, and Version 5.82 gives it the filename extension RAD). RESMAIN3 writes the results of the calculation to a set of files with the extension REP ("REPort"). The data input file is formatted in conformity with the FORTRAN NAMELIST input protocol, in which variables to be initialized in the program are listed by name in the input file and equated to the desired values. By preparing this file with the necessary names and values (a somewhat tedious undertaking) and adjusting the auxiliary file DIMENSON.DAT appropriately, a user can execute RESMAIN3 without invoking the GUI program.

Our recommendation is (1) that this launching mechanism be preserved in future versions of RESRAD, and that its relative independence of the GUI be maintained, so that the program can be launched directly from the command line or from a scripting program, without invoking the GUI front-end, and (2) that the procedure be documented so that users desiring to prepare the NAMELIST-formatted input file, make the modifications in DIMENSON.DAT, and run RESRAD from a script or wishing to run some preprocessing program on the input can do so. Primarily, the documentation should explain how each dimension value in the file DIMENSON.DAT is derived. It should explain the details of the auxiliary files KIFLG.DAT and KIFLG30.DAT (which are related to the decay chains). And it should define every variable in the NAMELIST-formatted input file, with units, and indicating conditions under which the variable is or is not used by RESRAD. There may also be other information that would be useful. This documentation could be printed in an appendix of the user's guide or it could be made available on the RESRAD web site.

We also recommend that DOE consider releasing the source code for RESRAD, making it available for downloading from a web site. We believe this change of policy would have three advantages: (1) Analysts using Unix workstations could recompile the code to function on their platforms, at least with command-line launching as we described in the previous paragraphs (having not seen the source code for the GUI, we do not know how difficult the conversion would be for that module). (2) Analysts with a good knowledge of programming can often resolve puzzling and subtle questions about what is being computed by referring to the source code. (This point is not intended to suggest that the developers do not support RESRAD and try to answer users' questions; as far as we know, the program is well supported.) (3) Experience seems to indicate that many useful suggestions for improving the program and the models it implements would come from programmers and analysts whose participation is currently precluded. In cases where there is particular concern about the authenticity of numbers imputed to RESRAD, it seems that some protocol could be developed that would require "final" or "official" results to be produced with a DOE-provided executable.

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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

"Setting the standard in environmental health"

Risk Assessment Corporation
417 Till Road, Newnan, SC 29107
Phone 803.536.4883 Fax 803.534.1995

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FINAL REPORT

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Radionuclide Soil Action Level Oversight Panel

October 1999

Contributing Authors

Jill Weber Aanenson, Scientific Consulting, Inc.
George G. Killough, Hendecagon Corporation
Kathleen R. Meyer, Keystone Scientific, Inc.
Arthur S. Rood, K-Spar, Inc.

Principal Investigator

John E. Till, Ph.D., *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*.

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 pre-designated 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 is to report the results of a sensitivity analysis conducted on the inputs and assumptions required for the use of RESRAD. 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 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 Monte Carlo interface has been developed and tested. The interface is now being 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 through the use of 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 are grouped into three categories: sensitive parameters, parameters with limited sensitivity, and

parameters not exhibiting sensitivity. The sensitive parameters identified in this fashion were the parameters for which uncertainty distributions have been developed. 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.

These parameters are given emphasis 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 are not sensitive or marginally sensitive have not been changed and are the same as those reported previously (DOE/EPA/CDPHE 1996). The only exceptions are 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* has determined that a different value is more appropriate based on the literature or site-specific data. *RAC* has 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 V. 5.82 are much higher than those predicted with older versions of the code. The single change in the formulation of area factor in the RESRAD code predicts 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 will be discussed in detail in a later section of the report, but the key characteristic of this new resuspension calculation is the use of site-specific data, namely soil and air concentration data.

SENSTIVITY 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).

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

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.

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

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.

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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.

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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

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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.

$$C_T = C_w \theta_w + K_d C_w (1 - (\theta_w + \theta_a))\rho_s \tag{4}$$

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
ρ_s = particle density (g mL^{-1})
K_d = distribution coefficient (mL g^{-1})

Assuming the total porosity is equivalent to the effective porosity, and relating the bulk density (ρ_b) to the particle density [$\rho_b = \rho_s(1-(\theta_a + \theta_w))$], Equation 4 can be solved for C_w giving

$$C_w = \frac{C_T}{\theta_w \left(1 + \frac{K_d \rho_b}{\theta_w}\right)} \tag{5}$$

The term, $1+K_d\,\rho_b/\theta_w$ represents the retardation factor (R). Solving to K_d yields

$$K_d = \frac{(R-1)\theta_w}{\rho_b} \tag{6}$$

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.

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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.

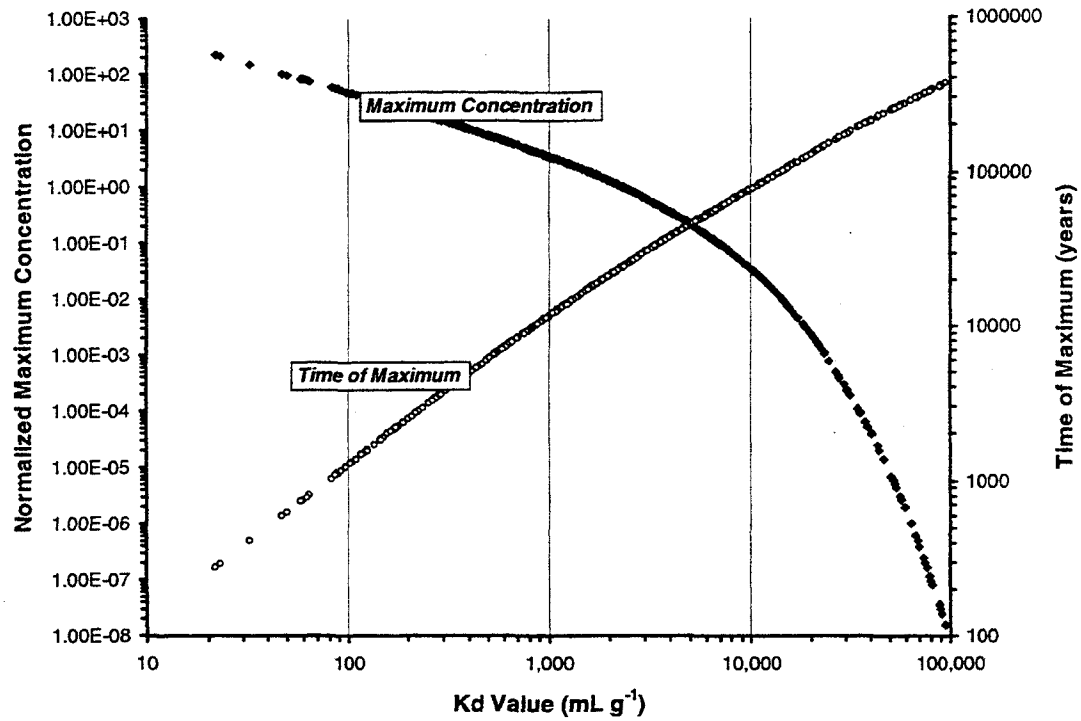

Figure 1. Normalized maximum concentration and time of maximum concentration as a function of the K_d value for ^{239}Pu. Maximum concentrations were normalized to the maximum concentration for a K_d of 2300 mL g^{-1}, which is the geometric mean of the distribution used in the analysis.

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.

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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

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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.

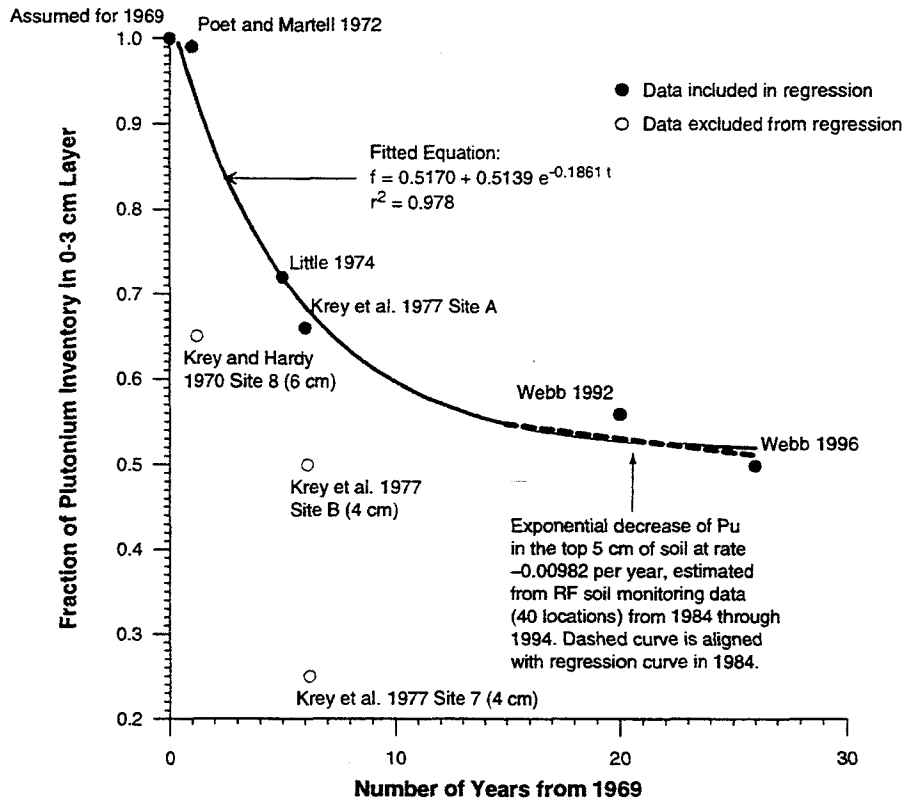

Figure 2. Regression curve fitted to ^{239}Pu data for the 0–3-cm layer of soil at Rocky Flats. The regression was based on data summarized by Webb (1996), which are plotted as black circles. The open circles were excluded from the regression. The dashed line represents an estimated exponential removal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

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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.

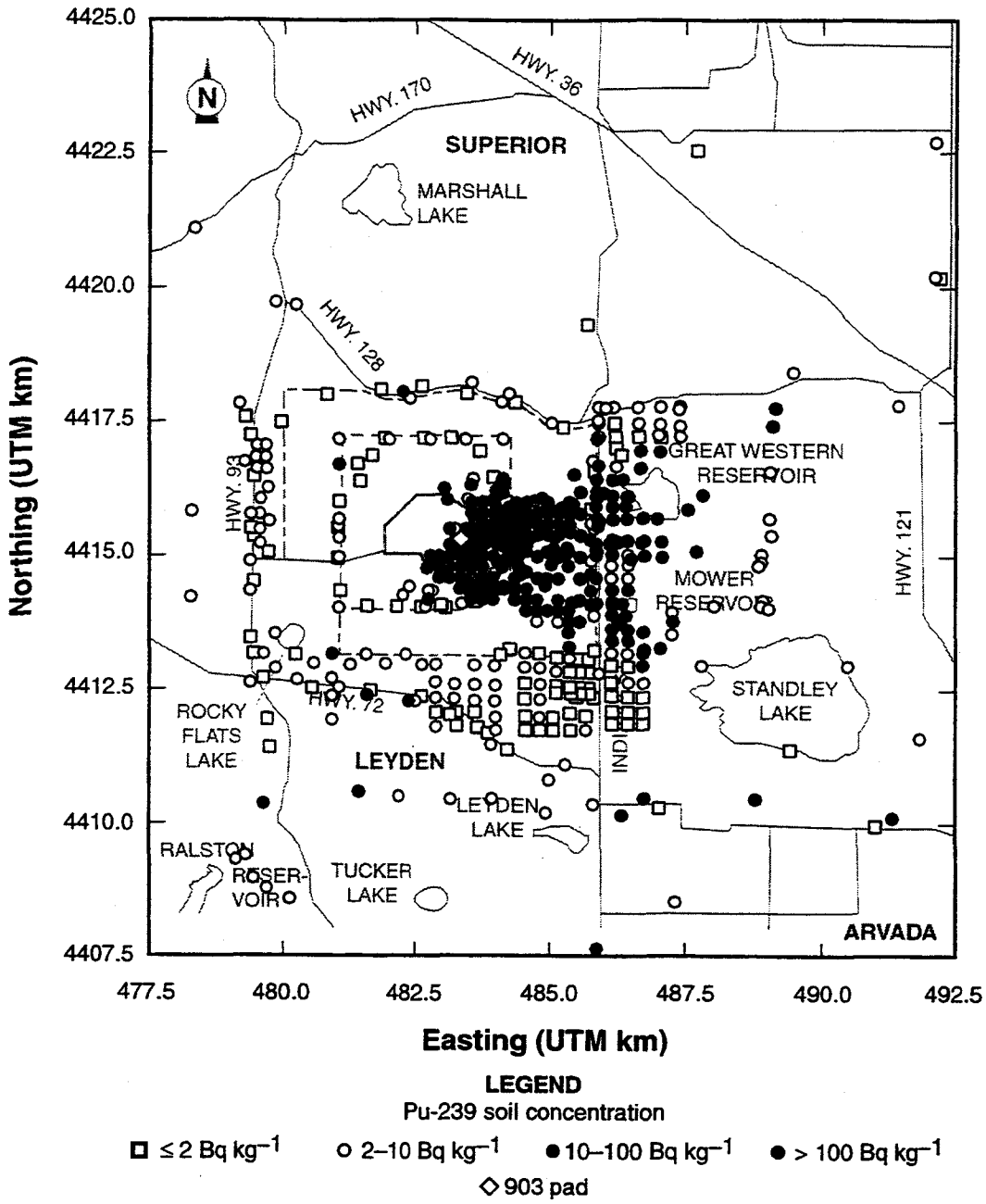

Figure 3. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model (100 Bq kg^{-1} = 2.7 pCi g^{-1}). The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), 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

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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.

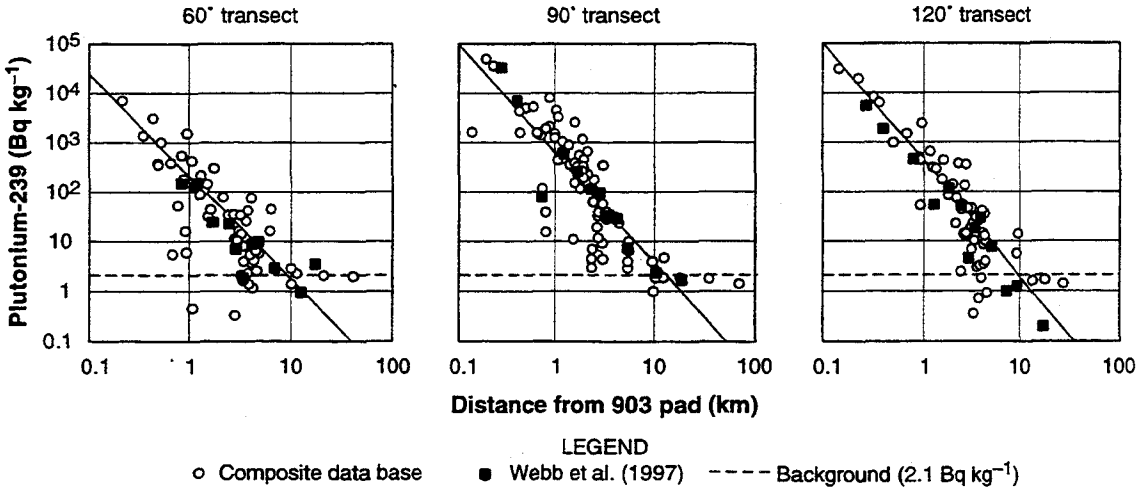

Figure 4. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 Area. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols. 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.

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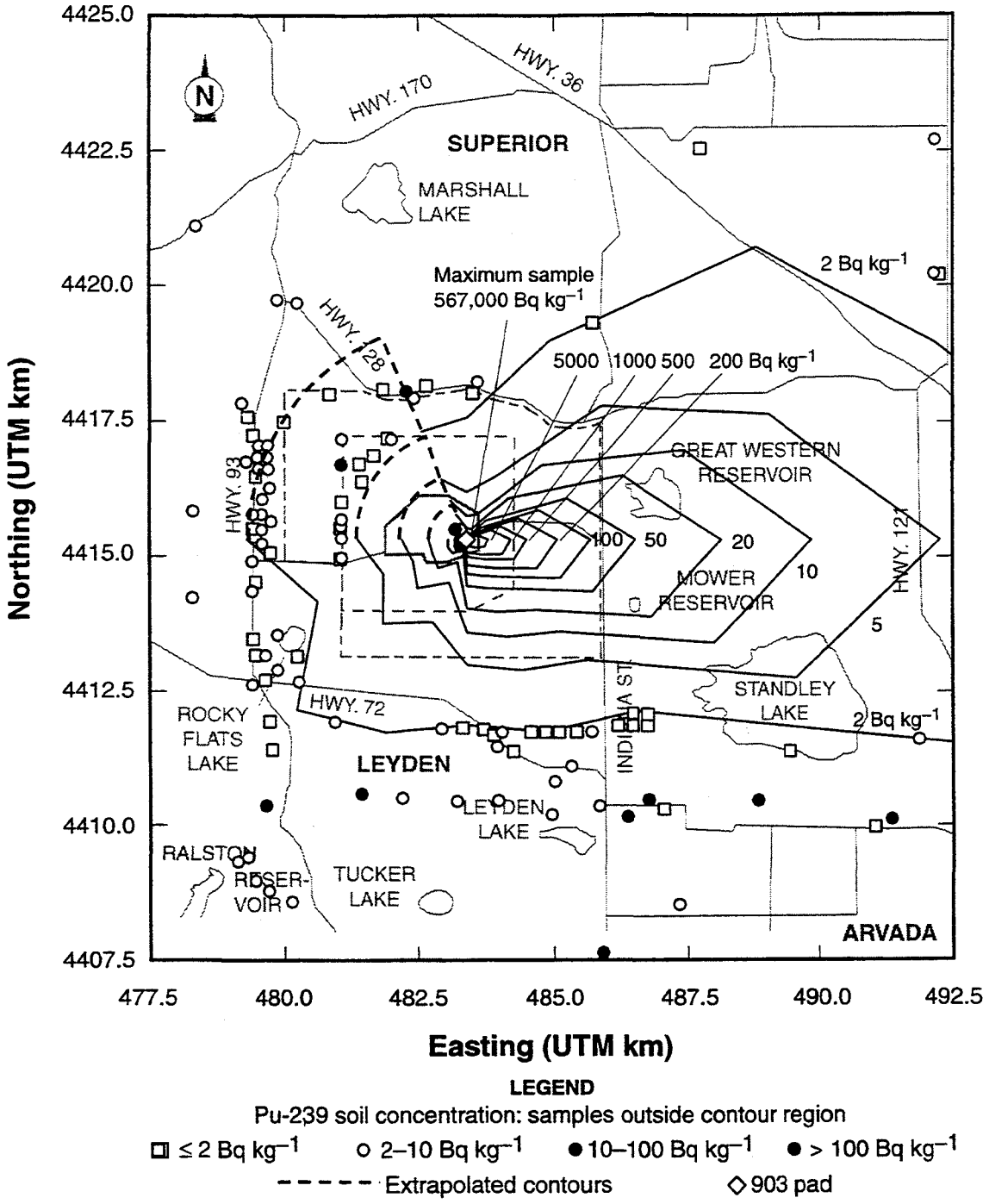

Figure 5. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text.

Dashed lines indicate extrapolation of the two northwest sectors from the 270° sector. Sample locations are shown outside the 2 Bq kg^{-1} (5.4×10^{-2} pCi g^{-1}) contour (approximately

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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

[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.

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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

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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 8½ 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

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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.

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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 heavy exercise	10.2 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 active	 8.8 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 Adult male, heavy activity	7.5 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 Range based on activity during day	11 7–12
Finley et al. (1994)	Age-specific distributions for chronic inhalation rates based on Layton (1993)	Adults 50th percentile Adults 5th, 95th percentiles	8.2 5.8, 11.6
EPA (1997)	Deterministic; Outdoor workers (15 men; 5 women)	Light activity Heavy activity	12 51
	Outdoor construction workers (19 males)	Light activity Heavy activity	24 34

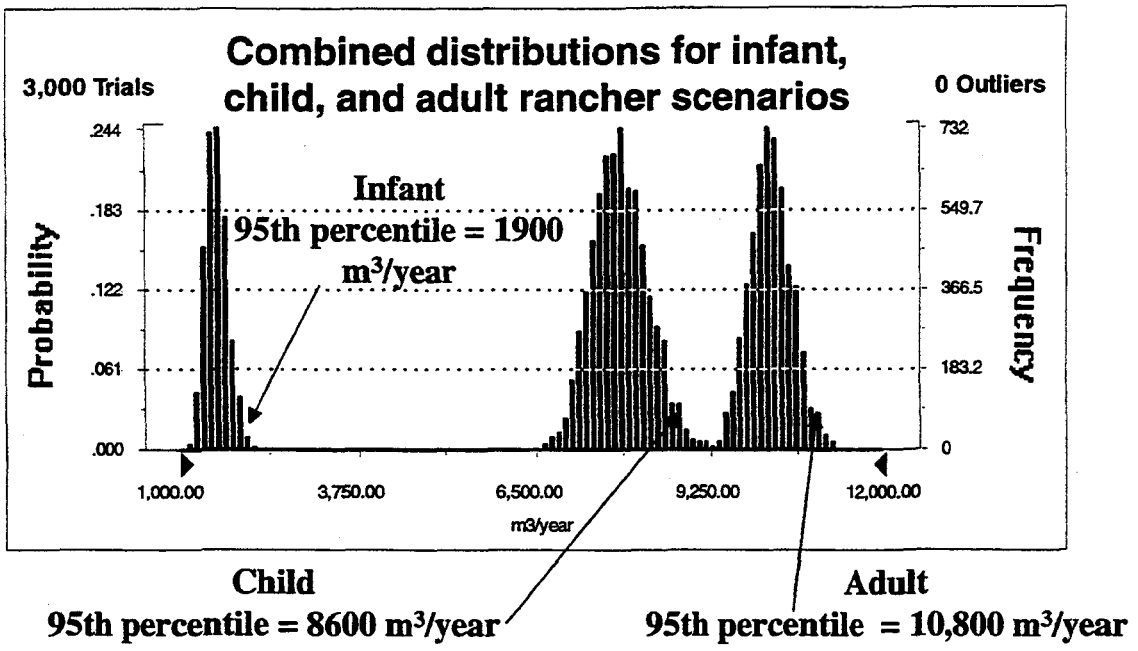

Figure 6. Distributions of breathing rates for the nonrestrictive scenarios: infant, child, and rancher. The 95th percentile of the distribution is shown for each scenario.

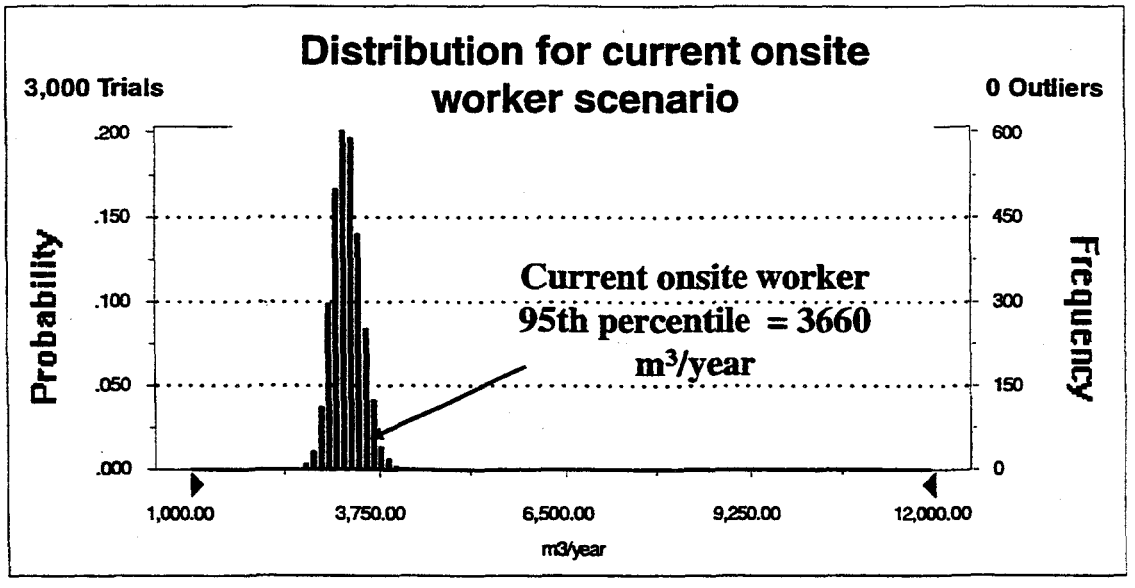

Figure 7. Probability distribution of breathing rate values for the restrictive scenario: current onsite industrial worker scenario. The 95th percentile of the distribution is 3660 $m^3\ y^{-1}$.

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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) NCRP Report 129 (NCRP 1999)	Scenarios based on literature review: Rural lifestyle (w/homes)—sparsely vegetated		
	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

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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.

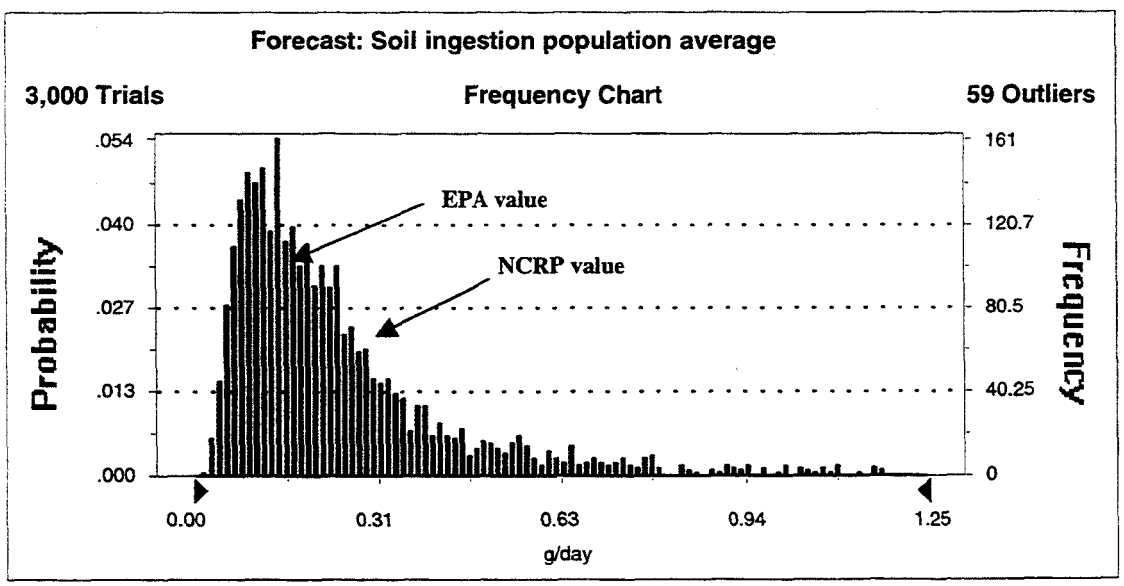

Figure 8. Frequency distribution of soil ingestion values from CrystalBall©. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation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

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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

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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 = 0.1255

	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})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
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})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
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	0.01740	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

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RAC Report No. 16-RSALOP-RSAL-1999-FINAL

FINAL REPORT

Task 5: Independent Calculation

Radionuclide Soil Action Level Oversight Panel

February 2000

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

"Setting the standard in environmental health"

Risk Assessment Corporation
417 Till Road, Neeses, SC 29107
Phone 803.536.4883 Fax 803.534.1995

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FINAL REPORT

Task 5: Independent Calculation

Radionuclide Soil Action Level Oversight Panel

February 2000

Contributing Authors

George G. Killough, Hendecagon Corporation
Arthur S. Rood, K-Spar, Inc.
Jill Weber Aanenson, Scientific Consulting, Inc.
Kathleen R. Meyer, Keystone Scientific, Inc.
Helen A. Grogan, Cascade Scientific, Inc.
Warren K. Sinclair, Warren K. Sinclair, Inc.

Principal Investigator

John E. Till, Ph.D., *Risk Assessment Corporation*

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

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GENERAL 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 (RFP) as a nuclear weapons research, development, and production complex. The RFP is located 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 (RSALs) developed for implementation by the DOE, the U.S. Environmental Protection Agency (EPA) and the Colorado Department of Public Health and Environment (CDPHE). RSALs are certain concentrations of one or more radionuclides in soil above which a criterion based on predicted radiation dose is exceeded. As a result of public concern about the soil action levels established in October 1996, 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.

This report, *Task 5: Independent Calculation*, presents the results of *RAC*'s independent assessment and describes the calculations and results of the soil action levels for seven exposure scenarios. The goal of radionuclide soil action levels is to protect people who may, in the near or distant future, come into contact with a site where the soil is contaminated with radionuclides at levels above background. Therefore, setting radionuclide soil action levels must consider the following:

- how particular radioactive materials are transported in the environment to people (transport pathways)
- how people might be exposed to the radioactive materials (exposure scenarios)
- how radiation dose to a person is assessed (radiation dosimetry)
- how radiation protection guidelines fit in (annual dose limits).

Because of these considerations, *RAC* focused on several factors important in the transport of radioactive materials in air and water in an area like Rocky Flats and developed exposure scenarios for the project. In designing the scenarios, *RAC* followed the principle that if the person living onsite full-time is protected, then the person living offsite will be protected. It was also important to understand the behavior of radionuclides in the soil and how soil can be disturbed or resuspended, because inhalation can be one of the important exposure pathways for those living on or near the site.

The exposure pathways considered in this analysis included inhalation, soil and food ingestion, and external irradiation. In addition, groundwater use for both irrigation and drinking water was assumed for some scenarios. The occurrence of a prairie fire that would remove the vegetative cover and result in increased resuspension of soil for a period of time was also considered.

The radionuclides ^{241}Am and the several isotopes of plutonium (^{238}Pu through ^{242}Pu) in the soil at Rocky Flats are the major radionuclides considered in the calculation. This contamination is not uniformly distributed across the facility, varying by more than a factor of 100. Uranium is also present in the soil at a few locations on the Rocky Flats site in concentrations above natural background, but uranium contamination resulted primarily from burn pits and isolated spills, and

the contamination is therefore more homogeneous in nature. For uranium, we assumed fixed isotope ratios for the ^{234}U, ^{235}U, and ^{238}U present at the site and expressed the composite uranium level in terms of a single isotope, ^{238}U. For plutonium, we calculated isotopic ratios and decay chains for all isotopes of plutonium, americium-241, and neptunium-237 starting with initial conditions measured in 1971.

For the calculations, we used the RESRAD Version 5.82, an updated version of the RESRAD program used for the earlier calculations, because it was the most practical choice and because we were required to make calculations with RESRAD in addition to any other code that may have been selected. To make the code better suit our needs, we designed extensions to RESRAD to include (1) consideration of the heterogeneity of radionuclide concentrations in soil around the site, (2) quantifying uncertainty in predictions of dose, (3) consideration of additional exposure scenarios, and (4) treating the possible occurrence of a large grass fire.

The modified approach for our analysis to include soil resuspension after a prairie fire accounted for the removal of the vegetative cover and increased resuspension of soil for a period of time. For each scenario, we incorporated the probability of a fire occurring in the area using fire statistics for this century in the Arapaho and Roosevelt National Forests and the Pawnee National Grasslands. For the plutonium assessment, the probability of a fire occurring on the rancher's land at the RFETS was estimated to be 1×10^{-3}.

Calculation of RSALs for uranium was done differently than that for plutonium because the nature and extent of contamination differed between the nuclides. Our treatment of plutonium considered a 10-km^2 contaminated area. Using spatially variable soil concentrations and measured air concentrations of plutonium around the site, we calibrated a suspension model so that the suspension rates of plutonium-contaminated soil would yield concentrations currently measured at the air samplers. This procedure was not extended to uranium because a) uranium-specific measurements were not available at the samplers and, b) uranium contamination is not as widespread as plutonium, and therefore would not be expected to respond in the same manner. Our investigation indicated that uranium was mainly limited to past disposal areas and burn pits. Furthermore, Litaor (1995) notes fundamental differences in solubility characteristics of plutonium and uranium that, in turn, affect their mode of dispersion in the environment.

The prairie fire was not considered for the uranium analysis because the smallest fire area considered in the fire statistics data set was 4.05×10^5 m^2, or 100 acres. Using the area encompassed by uranium contamination (100 m^2), yields a probability of a fire that is 5 orders of magnitude lower than that for the plutonium case. Additionally, only the inhalation pathway was affected by the fire and inhalation doses made up a small fraction of the total uranium dose. Nevertheless, we ran a trial fire case to verify that even if there were a fire, the doses from uranium would not be significantly higher. For this trial, we conservatively assumed that any fire occurring on the site encompassed a uranium-contaminated area.

Details of our technical approach for determining isotopic ratios, estimating concentration of plutonium in air, calculating uranium RSALs, calculating alternative groundwater dose from measurements in the literature, providing perspective on risk, and describing other computational details of the RSAL calculations are described in the report and in five appendices. We applied this approach to the Rocky Flats data using the most restrictive exposure scenarios approved by the Oversight Panel and assuming a 10% probability that the 15 mrem per year dose limit will be exceeded (i.e. a 90% probability that the dose limit will <u>not</u> be exceeded).

Using this approach, the technically derived RSAL for $^{239+240}Pu$ in soil at Rocky Flats would be 35 pCi g^{-1}. The results as presented are a reasonable indication of RSAL magnitudes based on purely scientific considerations. Table GS-1 shows the results of the plutonium calculations for each scenario at about the 10% probability level.

Table GS-1. Plutonium Soil Concentrations (pCi g^{-1}) at 10% Probability Level[a]

Scenario	Dose Limit[b]	
	15 mrem	85 mrem
DOE-1 (resident)	45	260
DOE-2 (open space)		6600
DOE-3 (office worker)		1600
RAC-1 (rancher)	35	
RAC-2 (child of rancher)	35	
RAC-3 (infant of rancher)	85	
RAC-4 (industrial worker)	90	530

[a] At the 10% probability level, there is a 90% probability that the dose limit will *not* be exceeded.
[b] Based on EPA guidance from 40 CFR 196; they are the dose limits used in the previous DOE/EPA/CDPHE

RSALs are also presented in this report for uranium isotopes (^{234}U, ^{235}U, and ^{238}U) for three scenarios: the DOE resident (DOE-1), the *RAC* rancher (*RAC*-1), and *RAC* child (*RAC*-2) scenarios. The DOE resident scenario was chosen for comparison with *RAC*'s methodology. The rancher and child scenarios were chosen because these scenarios yielded the most restrictive RSALs for plutonium. Assuming a viable groundwater pathway and a 10% probability, the technically derived ^{238}U RSAL for the most restrictive scenario (the rancher child) was 10 pCi g^{-1}.

We also developed an alternate method for calculating acceptable levels of radionuclides in soil. This method was based on calculating annual doses to the receptor for different remediation (i.e., cleanup) levels. The remediation level that resulted in a 10% probability that the 15 mrem dose limit would be exceeded defined the RSAL. This method more explicitly addresses the heterogeneity of the site and makes it possible to estimate RSALs that correspond more directly to a remediation strategy than does the sum-of-ratios technique used with RESRAD. The approach is more difficult to implement and therefore has not been fully automated in the analysis. However, because it is more explicit, it is a useful check on the sum-of-ratios method, and we include its results in these conclusions. This alternate calculation resulted in an RSAL at the 10% level of about 37 pCi g^{-1}, suggesting a value consistent with 35 pCi g^{-1} as a technically based RSAL for the Rocky Flats site.

While our methodology and the resulting RSAL values are scientifically defensible and are based on sound science, *RAC* believes that additional work could reduce some of the uncertainties and refine the RSALs. There were specific areas where more information or more organized research and scientific inquiry would have allowed us to make better parameter estimates. Foremost among these are data that quantify the impact of a prairie fire on the land now occupied by the Rocky Flats site and the data from the Actinide Migration Evaluation studies. Additional areas where research could enhance this work are described in this report.

by the Rocky Flats site and the data from the Actinide Migration Evaluation studies. Additional areas where research could enhance this work are described in this report.

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for Rocky Flats site. However, the final decision on setting the RSALs ultimately lies in the hands of the stakeholders, DOE, and other State and federal authorities. There are other criteria that influence the decision-making process for the Rocky Flats site, such as the cost of cleanup, protection of ecological resources, and community values. The approach to cleanup that is ultimately implemented by the DOE at the RFETS will involve many political, social, economic, and moral decisions. It is imperative that all involved in the decision process recognize these factors and the integration of ideas that must go into making a decision of this type.

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1. INTRODUCTION

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 facility under the name Rocky Flats Plant (RFP) as a nuclear weapons research, development, and production complex. The site is located 5–6 mi (8–10 km) from the cities of Arvada, Westminster, and Broomfield, Colorado and 16 mi (26 km) northwest of downtown Denver, Colorado. In this report, we refer to the facility either by the acronym RFETS or as the Rocky Flats site.

The current project evaluates the approach to soil action levels for radionuclides (RSALs) at Rocky Flats proposed for implementation by the DOE, the Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE). In response to public concern about the soil action levels proposed by these agencies (DOE/EPA/CDPHE 1996), DOE provided funds for the Radionuclide Soil Action Level Oversight Panel (RSALOP) to select a contractor to conduct an independent assessment and to perform an independent calculation of soil action levels for the RFETS. *Risk Assessment Corporation* (*RAC*) was selected to carry out the study. This report describes the calculation of these soil action levels based on an approach developed by *RAC*. The use of the RESRAD computer program (Yu et al. 1993; Chang et al. 1998) in the calculations was a contractual requirement.

The calculations are based on seven exposure scenarios that were described in the Task 3 report (Aanenson et al., 1999). Specified for each scenario is an annual limit for radiation dose to the resulting from exposure to Rocky Flats radionuclides. (By the term *subject*, we mean the hypothetical individual described by the scenario.) Environmental dose models are used to estimate dose from specified concentrations of radionuclides in environmental media. A radionuclide soil action level for a given exposure scenario is generally assumed to mean that concentration of the radionuclide in soil for which the model prediction of dose for the scenario subject equals the dose limit. Higher soil concentrations of the radionuclide would give dose predictions greater than the dose limit for the scenario, and lower concentrations would give dose predictions below the dose limit.

When multiple radionuclides are present in soil (say n of them), measured or hypothesized concentrations can be combined with the respective RSALs in a sum of ratios S:

$$S = \frac{\text{concentration}_1}{\text{RSAL}_1} + \cdots + \frac{\text{concentration}_n}{\text{RSAL}_n}$$

If S exceeds 1, the estimated dose produced by the combined observed concentrations exceeds the dose limit for the scenario. For ^{241}Am, ^{237}Np, and the several isotopes of plutonium (^{238}Pu through ^{242}Pu) in the soil at Rocky flats, the activity ratios may be assumed relatively constant over the domain of observation, although they change over time. Most of the radioactivity of these isotopes in the soil on and near the site came from waste stored in barrels on an unpaved pad in the 903 Area. Leakage from the barrels contaminated the soil beneath them, and the contamination was spread by wind-induced resuspension and deposition of soil particles. The redistributed contamination dominates other radioactivity in the soil over most of the site, and the spatially consistent isotopic ratios are attributable to the origin of these radionuclides from the waste barrels. The combination $^{239+240}$Pu is the most extensively measured quantity, and it has been the primary surrogate for plutonium and americium in the soil. It is possible to use the isotope ratios to express the maximum annual dose from americium and all plutonium isotopes as

a function of $^{239+240}Pu$ concentration in the soil. This relationship makes it possible to express a composite soil action level solely in terms of $^{239+240}Pu$ (although it depends implicitly on americium, neptunium, and the other plutonium isotopes, and their relative contributions to the maximum annual dose).

Uranium is also present in the soil at a few locations on the Rocky Flats site, in concentrations above natural background. The history of this contamination is different from that of the americium, neptunium, and plutonium from the 903 Area, and it does not appear possible to establish a simple spatial relationship between the uranium and plutonium-related isotopes. However, it is reasonable to assume isotope ratios for the ^{234}U, ^{235}U, and ^{238}U present at the site and thus to express the composite uranium level in terms of a single isotope, which we have chosen as ^{238}U. We present separate RSALs for plutonium and uranium isotopes.

The calculations reported here incorporate estimates of parameter uncertainty. Results for each scenario are presented in terms of the probability that the dose limit will not be exceeded, as a function of $^{239+240}Pu$, or ^{238}U, as the case may be. Uranium RSALs are based on the assumption of a small area of contamination (hot spot). Plutonium RSALs depend on a heterogeneous spatial distribution over a large area.

All calculations of soil action levels involve the use of RESRAD version 5.82. However, for the plutonium calculations, special techniques were required to circumvent the resuspension model that is programmed in RESRAD. Calculations external to RESRAD, reported in Section 4, establish relationships between levels of $^{239+240}Pu$ in the soil and atmospheric concentrations at primary locations of the scenario subjects. For the assumption of ground cover as it normally exists on the site, a regression analysis of air monitoring data for plutonium was carried out to estimate parameters for the resuspension model used in the external calculations. These calculations of air concentration at a receptor location are based on a smoothed representation of plutonium soil data for the site (Section 4) and integration of a Gaussian plume model over the contaminated source region (Section 5).

The possibility of catastrophic (or human-triggered) natural events cannot realistically be ignored. It is entirely plausible that a prairie-grass fire could burn all vegetation off large areas of the site, leaving bare soil for a year or more, with the potential for enhanced resuspension until the vegetation is reestablished. Scenario variants that assume the aftermath of an extensive fire thus require resuspension parameters for unvegetated soil. Parameters for such conditions are highly uncertain; our estimate of a resuspension factor from the literature has four orders of magnitude of uncertainty (Section 5.3). Such a loss of vegetation could also change the drainage characteristics of the soil until the natural growth was reestablished. One possible consequence is a change in the relative contamination of surface-water and groundwater, which could have an effect on dose estimates for some scenarios. Although we would expect any changes for the scenarios under study to be minor, this hypothesis has hydrological implications that cannot be explored within the resources of this project.

Some scenario variants discussed in this report assume the use of water from a contaminated aquifer. In general, transit times for plutonium from soil to groundwater exceeded the 1000-year time of compliance. Therefore, the water pathway had little impact on the results. Uranium isotopes are more mobile then plutonium isotopes, and groundwater doses from uranium were appreciable in the 1000-year compliance time frame. The results of these scenario variants must be considered tentative. Soil-to-water pathways cannot be treated definitively within this project because of their complexity and the incompleteness of data specific to this site. Some of these

questions are within the purview of the Actinide Migration Evaluation, and any treatment of them that might be attempted here would be premature. Instead, we have adopted for the *RAC* scenarios most of the water pathway parameters used for the DOE Resident scenario. That scenario considers only water-borne contamination through irrigation of garden crops from a well. DOE presumably assumed an uncontaminated municipal water source for all other water uses. For the *RAC* Resident Rancher scenarios, contaminated well water was assumed as the source for all water pathways including direct consumption. The RESRAD water-related transport parameters for the DOE Resident scenario were based on site-specific data (DOE/EPA/CDPHE 1996). Most of these parameter assignments were adopted for the water variants of the *RAC* scenarios. The exceptions were the soil-water partition coefficients K_d (mL g^{-1}), which are treated here as having order-of-magnitude uncertainty.

Calculations with the scenario variants that assume exposure to contaminated water indicate that in for uranium, the water pathway can dominate other exposure pathways. Specific cases are shown and discussed in Section 11. In addition, an alternative groundwater dose calculation for plutonium isotopes based on measured data is presented in Appendix B. But none of these results should be considered definitive in view of the incomplete information concerning radionuclide transport and exposure of subjects by these water pathways. The only conclusion to be drawn at this time is that the water pathways should not be dismissed out of hand. Rather, their potential for exposing people to radionuclides now residing in the soil on and near the Rocky Flats site should be investigated further as information from the Actinide Migration Project and other sources becomes available.

Initial indications, based partly on results with the DOE scenarios reported in DOE/EPA/CDPHE (1996), pointed to the inhalation of resuspended radionuclides as the dominant exposure pathway. Under the assumption of the normal ground cover for the site, however, the rank of this pathway in our calculations has diminished in favor of soil ingestion. In the case of uranium, the contaminated water pathways (for which the caveats of the previous paragraph must be kept in mind) show a potential for dominant importance.

However, there are two other reasons for the change in pathway rank:

(1) The change in resuspension modeling
(2) Replacement of the radiation dose coefficients used for the analysis reported by DOE/EPA/CDPHE (1996) with those currently recommended for the public by the International Commission on Radiological Protection (ICRP).

Updating the dose coefficients had the effect of simultaneously reducing the annual dose per becquerel intake by inhalation and increasing the dose per becquerel intake by ingestion. Thus, the change in the dose coefficients increased the ingestion dose and decreased the inhalation dose from resuspended radionuclides, leaving the ingestion dose the dominant one for some cases.

In the variants of *RAC* scenarios that consider the aftermath of a fire, however, the generally high rate of resuspension restores the inhalation dose to its position of dominance, although depletion of the surface soil via leaching complicate the picture. When small K_d values occur in the Monte Carlo sampling in conjunction with sufficiently small resuspension factors, dose components from the ingestion pathways can dominate. Such events have a sufficient probability to influence the curves that show the probability of exceeding the dose limit as a function of radionuclide concentration in soil (Section 11).

The probability curves in Section 11 provide a compact means of appraising the relationship between the annual dose limit for a scenario and radionuclide levels that would (according to the environmental model) produce that annual dose. The relationship is affected by uncertain parameters, and this component of uncertainty is taken into account by estimating the probability that the annual dose limit will be exceeded. This probability is plotted against the concentration of $^{239+240}Pu$ on the horizontal axis.

This kind of analysis facilitated by these plots provides information to help interested parties quantify standards for acceptance of soil action levels. It does not provide value judgments about what probability criterion (e.g., 5% or 20%) should be adopted in a given case. However, there is regulatory precedent for 10% (Section 10). The analysis provides RSALs for several exposure scenarios, but it cannot settle the question of which scenarios should be judged most relevant to remediation of the site. Land use, and institutional control, environmental tradeoffs, and cost are matters that involve political considerations beyond our scope. Our information bears on such questions, but the values of the parties involved are required for deciding them. Weight must also be given to practical constraints of what is technically feasible.

We wish to emphasize that the calculations described in this report demonstrate a general approach to the question of Rocky Flats soil action levels. It has not been possible, within the resources of this project, to consider some refinements that would improve the assessment, but we believe the discussions and results adequately demonstrate the approach. We applied this approach to the Rocky Flats data using the most restrictive exposure scenarios approved by the Oversight Panel and assuming a 10% probability that the 15 mrem per year dose limit will be exceeded. The results as presented are a reasonable indication of RSAL magnitudes based on purely scientific considerations. Recommendations for further research are presented. If these recommendations are implemented, some RSALs could change.

2. BACKGROUND OF THE STUDY

The focus of the current project was to develop a methodology for determining radionuclide soil action levels (RSALs). In 1996, the DOE, the Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE) proposed interim radionuclide soil action levels to be used in the cleanup of the Rocky Flats site (DOE/EPA/CDPHE 1996). As a result of public concern about the soil action levels established in October 1996, the Radionuclide Soil Action Level Oversight Panel (RSALOP) was formed. The RSALOP was a group of community members with considerable experience in Rocky Flats issues. In 1998, DOE provided funds for the RSALOP to select a contractor to conduct an independent assessment of the proposed interim radionuclide soil action levels (RSALs) and to calculate RSALs for the RFETS. Through a competitive bidding process and evaluation, *Risk Assessment Corporation* (*RAC*) was selected by the RSALOP to carry out the study. Work began in October 1998 and was completed in March 2000.

To understand the scope of this report, it is important to understand the design objectives of the entire project. These objectives emerged from the scope of work and determined much of this analysis' direction. Key design objectives are listed below.

1. Base the soil action level on a dose constraint rather than a level of risk.
2. Consider two dose constraints: 15 mrem (0.15 mSv) in a year for unrestricted use of the site and 85 mrem in a year for restricted use. The dose limits were those chosen for the 1996 assessment (DOE/EPA/CDPHE 1996) and are based on EPA Draft Title 40 CFR 196, which states that a remediation standard of 15 mrem y^{-1} should be used at sites with radioactive material in all environmental media (EPA 1996). The radiation dose to be received by an unrestricted release exposure scenario will not exceed 85 mrem y^{-1} so that any individual will not receive more than the ICRP recommended dose limit of 100 mrem even if land use restrictions fail in the future (ICRP 1977).
3. Consider any realistic scenarios of exposure for the future and do not be limited to using scenarios that had previously been proposed.
4. Include uncertainties in the calculation to the greatest extent possible.
5. Incorporate site-specific data into the calculation where they are available.
6. Evaluate different computer codes that are available for calculating RSALs and select one to use that is the best for the situation at Rocky Flats. The RESRAD environmental transport computer code, Version 5.61 (Argonne 1993), was used in the previous assessment as specified by DOE Order 5400.5.
7. Use a documented and reviewed computer code; however, modify this code if possible to improve the quality of the calculation.
8. Evaluate all input parameters to the RESRAD computer code and suggest alternatives if values are not appropriate for the Rocky Flats site.
9. Complete the work within the time constraints given and interact with the RSALOP and the public at monthly availability sessions and formal meetings.

This study developed out of concern about the methodology and lack of public input involved in the process of establishing interim soil action levels by DOE/EPA/CDPHE (1996). These radionuclide soil action levels or cleanup levels are presented in Tables 2-1 and 2-2. Table 2-1 shows the DOE/EPA/CDPHE individual soil action level results. Table 2-2 shows the sum-

of-ratios example that considered a fixed ratio of ^{239}Pu to ^{241}Am. Comparisons between these results and the results of the *RAC* calculations to be presented in this report are discouraged because the two sets of calculations were performed with (1) different dose conversion factors and (2) different resuspension models and data. Additionally, (3) in the DOE calculation, the principal pathway was inhalation; in the corresponding *RAC* estimate, it was ingestion, and (4), the DOE calculation was deterministic, whereas examples of RSAL numbers using the *RAC* methodology presented here represent the 90th percentile of a stochastic simulation. *RAC* also included the effects of a prairie grass fire on the calculation of soil action levels for every scenario

Table 2-1. Individual Radionuclide Soil Action Levels (in pCi g^{-1}) Proposed by DOE/EPA/CDPHE in October 1996 [a]

	Resident		Office worker	Open space user
Radionuclide	15 mrem[b]	85 mrem[b]	15 mrem[b]	15 mrem[b]
Americium-241	38	215	209	1283
Plutonium-239,240	252	1429	1088	9906
Uranium-234	307	1738	1627	11500
Uranium 235	24	135	113	1314
Uranium-238	103	586	506	5079

[a] Taken from Table ES-1, DOE/EPA/CDPHE 1996.
[b] Annual dose limits

Table 2-2. DOE/EPA/CDPHE Example of Radionuclide Soil Action Levels (in pCi g^{-1}) Based on the Sum-of-Ratios

	Resident		Office Worker
Radionuclide	15 mrem	85 mrem	15 mrem
Americium-241	21	117	101
Plutonium-239,240	115	651	562

This project was broken into eight tasks. Two of the tasks, public interaction (Task 7) and interaction with the actinide migration evaluation (Task 8) occurred throughout the course of the project and impacted the evolution and outcomes of other task reports. This Task 5 report builds on the groundwork laid by three previous reports: Tasks 1, 2, and 3.

The first task of the study (Task 1, Cleanup Levels at Other Sites) was designed to provide the RSALOP with an unbiased evaluation and comparison of previously developed soil action levels for the RFETS and other facilities (Weber and Till 1999). Soil action levels and other cleanup criteria have been established at a number of national and international sites. Based on our review of soil action levels at other sites, *RAC* concluded that the soil action levels developed by the DOE/EPA/CDPHE for use at the RFETS are significantly higher than action or cleanup levels at other facilities. This was the case even when normalized to dose, i.e., presented as soil level per unit dose. *RAC* was able to identify the differences between levels in almost every case. The discrepancies were always related to different parameter valuation or different baseline assumptions.

In Task 2, we evaluated five environmental assessment computer programs for use in the project (RESRAD, MEPAS, GENII, MMSOILS, and DandD) (Killough et al. 1999). Based on this evaluation, *RAC* selected an updated version of the RESRAD code, Version 5.82, for all independent calculations of soil action levels for the current project. RESRAD Version 5.82 employed a revised methodology for resuspension that *RAC* found unsatisfactory for evaluating Rocky Flats soil action levels. This led us to devise our own treatment of resuspension, which is described in this report.

The outcome of the RESRAD calculation is strongly controlled by a few parameters, as shown in the Task 3 report. The controlling parameters are mass loading (resuspension), soil-water equilibrium distribution coefficient, mean annual wind speed, and area of the contaminated zone. *RAC* studied the influence of these and other parameters on determining the soil action levels. The input parameters to RESRAD were described in detail in the Task 3 report (Aanenson et al. 1999). Each parameter of significance to the calculation was described, and distributions of values for significant parameters used in the independent calculation were given. Many other parameters are discussed in the Task 3 report, not because they significantly impacted the calculation, but because we changed the values from their DOE/EPA/CDPHE value to better reflect the current state of knowledge.

Another important consideration is the dose conversion factors (DCFs), which are the radionuclide-specific factors that determine the dose per unit concentration of inhaled or ingested radionuclide. We used DCFs from the most recent ICRP reports (67 and 71) addressing the subject (ICRP 1996) rather than the values from ICRP 30 (ICRP 1979) used in the original DOE assessment. The newer ICRP 71 (ICRP 1996) inhalation dose coefficients for plutonium are lower than those reported in ICRP 30 primarily because the newer respiratory tract model assumed a reduced uptake of plutonium from the lung. The newer model also considers dose to specific cells at risk (target cells) rather than calculating an average dose over a region.

We also studied some important scenario-related parameters in detail, such as the breathing rate and soil ingestion rates (Aanenson et al. 1999). The exposure scenarios are an integral part of the soil action level work, and *RAC* invested considerable thought and time to ensure the scenarios would be protective of people who may come into contact with the site in the future. Each scenario hypothesized the exposure characteristics of a single individual, with a defined set of behaviors and physical attributes (i.e., exposure scenarios were treated deterministically in this analysis). *RAC* evaluated the three scenarios described in the existing soil action level report (DOE/EPA/CDPHE 1996) and developed four additional scenarios after numerous discussions with the RSALOP. The scenarios are defined by numerous parameters of varying importance. Examples of important 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 focused our greatest effort on establishing values for breathing rate and soil ingestion because they were parameters in which the panel expressed primary interest.

RAC also developed a Monte Carlo interface for RESRAD to estimate uncertainty distributions for the final dose and soil action level values for each of the scenarios and used probability distributions developed for the input parameters (Aanenson et al. 1999). This interface also helped us consider the nonuniform spatial distribution of plutonium and americium in the soil on and near the RFP site. The interface was calibrated to reflect site-specific conditions and used site-specific historic data, particularly air monitoring and soil concentration data.

This current report builds on the assumptions and methodologies explained in the reports for Task 1 (Weber and Till 1999), Task 2 (Killough et al. 1999), and Task 3 (Aanenson et al. 1999). Therefore, we encourage the reader to review these previous reports for specific details about the techniques we used to estimate soil action levels and input data used in our analyses.

3. ISOTOPIC RATIOS FOR PLUTONIUM, AMERICIUM, AND NEPTUNIUM

Plutonium radioactivity measurements in the soil and air at Rocky Flats are expressed as the sum of the isotopes ^{239}Pu and ^{240}Pu. Other isotopes of plutonium are present in addition to americium and neptunium. Relative amounts of these radionuclides change with time according to their initial proportions and their radioactive decay schemes. The dominant signal over most of the site is radioactivity from solvents leaked onto the 903 Area from storage barrels in the 1960s and dispersed by wind-driven resuspension of soil particles. Thus, ratios of the radionuclide activities are predictable and are assumed approximately uniform over the site.

In the calculations of soil action levels, we considered ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, ^{241}Am, and ^{237}Np. Krey et al. (1976) summarized measurements of mass ratios of the plutonium isotopes made in 1971. Table 3-1 normalizes these ratios to a total of 100 g of plutonium and shows the corresponding specific and absolute activities of the isotopes. Table 3-1 also shows quantities of ^{241}Am and ^{237}Np calculated from the $^{241}Pu \rightarrow {}^{241}Am \rightarrow {}^{237}Np$ decay chain for the year 1971. The calculation assumed that a unit of ^{241}Pu activity was present in 1965, with no decay products present. We then adjusted the quantities calculated for 1971 to make ^{241}Pu agree with the activity level shown in the table, giving the correct relative proportions of the decay products for that year.

Table 3-1. Initial (1971) Isotope Ratios for Plutonium and its Decay Products

	Specific activity		Mass	Activity	
Isotope	TBq g^{-1}	pCi g^{-1} [a]	g	TBq	pCi
Pu-238	6.34×10^{-1}	1.71×10^{13}	6.79×10^{-3}	4.31×10^{-3}	1.16×10^{11}
Pu-239	2.30×10^{-3}	6.22×10^{10}	9.49×10^{1}	2.18×10^{-1}	5.90×10^{12}
Pu-240	8.43×10^{-3}	2.28×10^{11}	4.84	4.08×10^{-2}	1.10×10^{12}
Pu-241	3.81	1.03×10^{14}	2.19×10^{-1}	8.34×10^{-1}	2.26×10^{13}
Pu-242	1.45×10^{-4}	3.92×10^{9}	1.36×10^{-2}	1.97×10^{-6}	5.33×10^{7}
Am-241	1.27×10^{-1}	3.43×10^{12}	7.26×10^{-2}	9.22×10^{-3}	2.49×10^{11}
Np-237	2.64×10^{-5}	7.14×10^{8}	3.55×10^{-4}	9.38×10^{-9}	2.54×10^{5}

[a]The units pCi g^{-1} are used almost exclusively throughout this report. Although these units are not the ones commonly used among experts, they are the most readily recognizable to the RSALOP. To convert from pCi g^{-1} to Bq kg^{-1}, multiply the pCi g^{-1} quantity by 37.

Figure 3-1 shows the behavior of the isotopes in Table 3-1 over time. Plutonium-241 decays by beta to ^{241}Am, which we estimate will reach a maximum in 2032. Americium-241 decays by alpha to ^{237}Np. Plutonium-239, ^{240}Pu, and ^{242}Pu have half-lives of thousands of years and undergo negligible radioactive decay during the period considered. We included the effect of leaching with a soil-water partition coefficient $K_d = 2000$ mL g^{-1} for all plutonium isotopes and $K_d = 1000$ mL g^{-1} for americium. This would simulate some removal of these radionuclides from surface soil.

The initial conditions were based on the relative activities in 1971 given in the table. The kinetic behavior of the decay chain $^{241}Pu \rightarrow {}^{241}Am \rightarrow {}^{237}Np$ was calculated by standard methods. The calculation predicted a maximum for ^{241}Am in the year 2032. Krey et al. (1976) made a

similar estimate. The radioactive decay products of other species are not shown, but these radionuclides form in quantities that are negligible for dose.

In simulations involving RESRAD, we began all calculations in the year 1971 to fully account for the kinetics of the species. RESRAD makes the decay chain calculations, in addition to simulating removal of radioactivity from surface soil over time. We estimated initial concentrations for 1971 by back calculating from the desired levels of $^{239+240}Pu$ in the year 2000 (or 2100) based on scenario assumptions. Uncertainties enter through the assumed soil level of $^{239+240}Pu$ in 2000 and through other parameters.

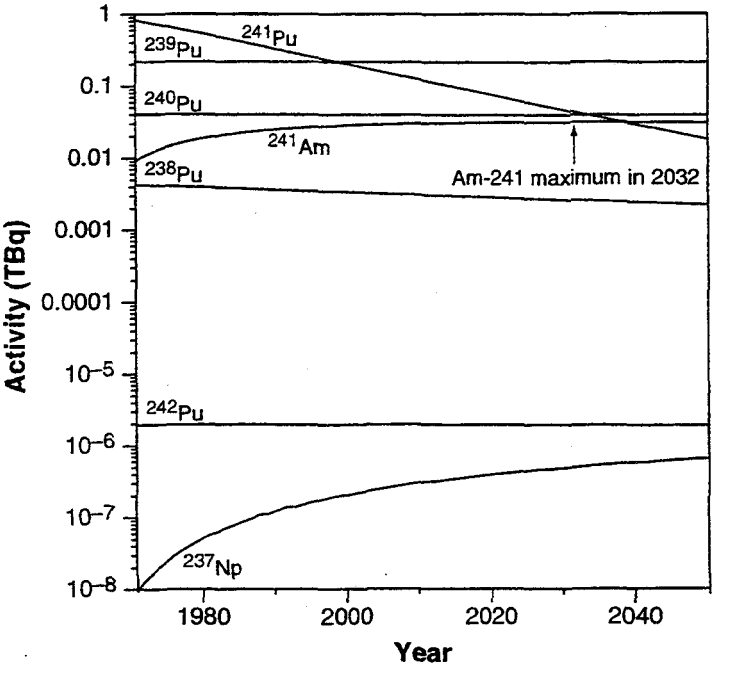

Figure 3-1. Relative kinetics of plutonium, americium, and neptunium in Rocky Flats soil from 1971 to 2050. The activity proportions correspond to 100 g of plutonium with isotope mass ratios given by Krey et al. (1976).

4. RECENT SPATIAL DISTRIBUTIONS OF PLUTONIUM IN THE SOIL NEAR THE ROCKY FLATS SITE

Concentration of radionuclides in soil at Rocky Flats is not uniform across the site. Numerous historic and recent studies have measured concentrations and spatial variation of plutonium in soil. We have used these studies to compile a composite database of soil concentrations for $^{239+240}Pu$, indexed by their distance and angle from the 903 pad[a], where the highest measured levels exist. Section 4.1 describes a model that approximates the spatial distribution of $^{239+240}Pu$ on and near the Rocky Flats site. The model is based on the composite database and provides a means of taking the nonuniformity of the concentration into account in estimating exposure of scenario subjects to plutonium.

Uranium isotopes are a different matter. Soil concentrations of uranium that are above natural background are confined to a relatively few small areas on the Rocky Flats site. Analysis of these "hot spots" can be accomplished with straightforward applications of RESRAD. Section 6 of this report discusses the assessment of uranium.

4.1 A Spatial Model of the Plutonium-239 Concentration in Soil

A serious complication in applying RESRAD to Rocky Flats is the inhomogeneous spatial distribution of plutonium in the soil. RESRAD works with a specified region within which the soil concentration is mathematically treated as being uniform, although the developers relax that assumption to accept variation within a factor of three. However, plutonium concentration in the soil increases by a factor of more than 100 from Indiana Street westward to the 903 Area. Thus, it is not possible 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 factor-of-three test for homogeneity.

Our approach to estimating soil action levels requires that we recalculate RESRAD's ratio of air and soil radioactivity concentrations, in such a way that they account for the large spatial variations in soil radioactivity. In effect, it is necessary to estimate realistic air concentrations of radioactivity and use them to modify the parameters in RESRAD. Such an approach requires a model of the spatial distribution of $^{239+240}Pu$ radioactivity concentration in soil on and near the site. We use the term model because we need an estimate of the concentration for any specified location, including locations where no measurements have been made. Moreover, the derived distribution should be smoothed somewhat, leveling the considerable scatter in the data. Otherwise, attempts at numerical integration with the model would run into difficulty, and numerical integration is necessary to add the contributions to resuspended plutonium from different locations.

To define such a model, we need to begin 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 set of

[a] The 903 pad is alternately referred to as the 903 Area and the 903 pad throughout this document. The two phrases represent the same thing. We generally refer to the 903 Area when we are discussing the plutonium contamination, because that contamination came from the area originally known as the 903 Area. The 903 pad refers to the asphalt pad placed over the area during cleanup of the disposal site. This is the phrase generally used when discussing research that took place after cleanup of the area.

measurements that did not meet these criteria is discussed below). The sampling depth is of particular importance because recent field and theoretical work reported by Webb et al. (1997) has established a parametric depth profile for ^{239}Pu at Rocky Flats that can be applied generically to adjust samples taken to various depths to a common basis.

In general, we follow the example of Webb et al. (1997) and use the ^{239}Pu concentration in the 0–3 cm layer as representative of the concentration of plutonium in resuspendable soil. The generic profile indicates that essentially all plutonium in the soil at Rocky Flats is currently confined to a depth of 20 cm, with concentration that decreases with increasing depth. Concentrations based on samples taken to depths less than 20 cm can be adjusted to the 0–3 cm depth by hypothesizing a profile for the sample that is proportional to the standard one of Webb et al. (1997), thus extrapolating to depths greater than 20 cm.

Evolution of the profile over time is less clear. After its initial wind-borne transport from the 903 Area, it appears that plutonium migrated within a few years into the soil where it was deposited and established the 20-cm profile. Krey and Hardy (1970) indicated that plutonium measured in 1969 and 1970 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 pad and in locations where soil had been disturbed. However, Krey (1974) subsequently defended the conclusion of Krey and Hardy (1970).

The downward migration seems to have been rapid and to have attenuated quickly, and there has been no clear indication of migration deeper than 20 cm. This kind of schedule is qualitatively supported by estimates of the inventory of soil plutonium summarized by Webb (1996). These estimates are consistent with a regression curve showing an initial exponential removal of 40 percent of the inventory out of the 0–3 cm layer within 10 years (Figure 4-1). The curve indicates an asymptotic level of about 50% of the initial deposition remaining in the 0–3-cm layer. This schedule seems too gradual to be consistent with Krey and Hardy (1970) and Krey (1974) and with some observations of Krey et al. (1977). Data from some of the locations sampled in these studies were omitted from the regression because of the apparently inconsistent interpretations. These omitted observations are presented as open circles in Figure 4-1. Rood and Grogan (1999) give a fuller discussion of the questions involved.

To provide additional support for our choice of data used in the regression, we have shown the number of sampling sites that define each point on the graph (Figure 4-1). Note that the number of sites sampled by Webb and Little far exceeds the number of sites represented by Krey and others. Had the data from each individual site been readily available, it might have been worthwhile to plot each entire data set and perform the regression using the aggregated data. But we believe we would have found a very similar curve had we had done this.

Results from these studies are perplexing. There appears to be a clear evidence of a decrease in the 0–3-cm plutonium inventory between 1972 and 1989 based on the work of Little (1974), Webb (1992), and one sampling site in Krey et al. (1977). However, two of the other sites measured by Krey et al. (1977) show substantially less plutonium in the surface (0–5 cm) than was observed by Webb and Little. Little (1976) measured depth profiles at 10 sites, and Webb (1992) resampled these same sites in 1989, while Krey's later measurements were from only three sites.

Numerous processes can influence plutonium migration in the subsurface, and these processes are both temporally and spatially variable. These processes include soil erosion (Webb 1993); colloidal movement (Bates et al. 1992); biotic perturbation (Litaor et al. 1994; Winsor and Whicker 1982); and soil cracking (Higley 1994). These processes are not well understood and are currently an area of research at the RFETS. 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 time. Most of the time, plutonium has migrated very little.

We do not doubt the accuracy of the work Krey and his coworkers performed in the 70s, and we think it is likely that depth distributions will vary among locations. Krey's data certainly suggest large variability both spatially and temporally. The regression equation is simply an empirical means to summarize the gross behavior of plutonium in the soil as indicated by a variety of data sets.

It is likely that natural processes continue to remove plutonium from the surface soil. In addition to deriving the regression curve, we performed a statistical analysis on $^{239+240}Pu$ samples from the 0–5-cm depth that were taken as part of the Rocky Flats monitoring program. These samples were taken annually from 1984 through 1994, at 40 locations, with distances roughly 1 mile (1.6 km) and 2 miles (3.2 km) from the center of the site and at direction intervals of 18°. Using the aggregated data, we estimated a loss rate of approximately 1 percent per year during the 11-year sampling period (Figure 4-1). Despite considerable scatter in the data, separate estimates based on the inner and outer rings of sample locations were consistent, giving nearly identical values of the rate coefficient. An 85% confidence upper bound for the rate coefficient is -3.02×10^{-4} per year, excluding zero at this confidence level (thus we may conclude that the rate coefficient is negative, corresponding to a plutonium loss that is detectable at the 85% confidence level). Ninety-percent and higher confidence levels give positive confidence upper bounds and thus do not rule out zero. Figure 4-1 indicates that the rate estimate gives a good approximation to the slope of the regression curve during the relevant period (dashed segment).

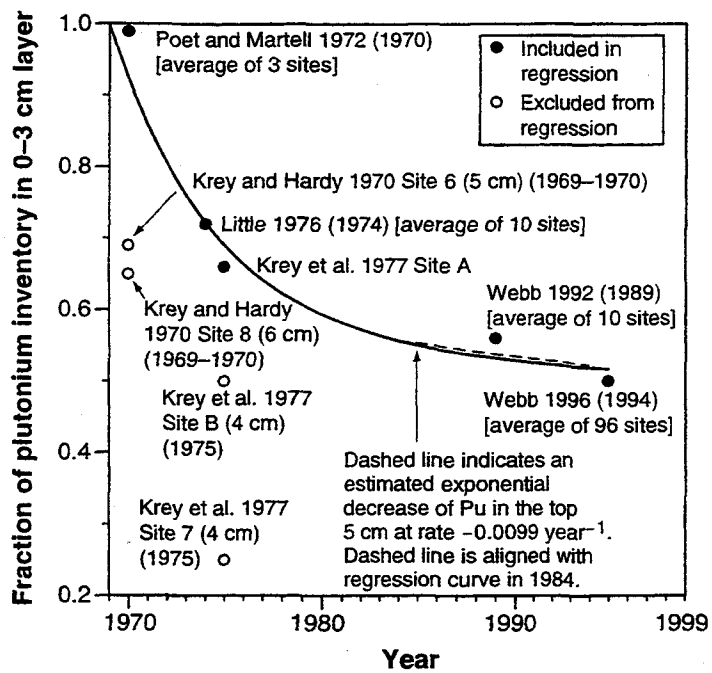

Figure 4-1. Regression curve based on data summarized by Webb (1996). The curve represents the equation $y = a\exp(-b_1 t) + (1-a)\exp(-b_2 t)$, with nonlinear least squares–determined parameters $a = 0.54769$, $b_1 = -0.00255$, and $b_2 = -0.1825$. The black circles represent the data used for the regression, and the white circles indicate omitted observations. The dashed line refers to a separate analysis. It is based on plutonium data from the 0–5-cm layer at 1-mile and 2-mile distances from the 903 pad (20 locations at each distance), representing the years 1984–1994. The estimated exponential removal rate is approximately 1% per year. We find that the rate is negative at the 85% confidence level, but not at 90% or higher confidence levels.

The raw data for the plutonium database were obtained from two sources:

1. A computer archive of 1122 results of soil samples, deposited with the Colorado Department of Public Health and Environment (CDPHE) by M. Iggy Litaor.[b]

2. Table I-2 of Appendix I from Ripple et al. (1994).

The archive (1) of Litaor/CDPHE provided Colorado State Plane (CSP) coordinates (ft) and activity concentrations (pCi g^{-1}) for observations reported by Illsley and Hume (1979). It also provided the CSP coordinates for the 40 locations of the RF monitoring series mentioned previously (rings at approximately 1 and 2 miles 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; the plutonium results for these locations were taken from the 1994 environmental monitoring report (RFETS 1994) rather than from the archive. Many of the data in the Litaor/CDPHE archive could not be documented and therefore were not used. One series, with code numbers PT000–PT124, however, was considered essential because of the coverage that it

[b] This data archive is available from the Colorado Department of Public Health and Environment (303-692-2000)

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provided near the 903 pad. The Rocky Flats sampling protocol specified a sampling depth of 0–5 cm, and we have assumed that all observations in the PT series were taken in conformity with this protocol, but it is possible that the series contains some values that are based on shallower depths. No other data from this archive were used.

The compilation of Ripple et al. (1994) (item 2) provides good documentation and discussion of a variety of measurements taken during 1969–1971. The protocols vary, and sampling depths range from 1 cm to 20 cm. The plutonium activity is reported as mCi km^{-2}, which we converted to Bq kg^{-1} using an assumed average bulk soil density of 1 g cm^{-3}. We used the raw data as presented and not the numbers in the column labeled "corrected." Coordinates in the appendix of Ripple et al. (1994) were given in the Universal Transverse Mercator (UTM) system (m). We note that the Litaor's/CDPHE archive (item 1 listed above) included the data from Ripple et al. (1994) and was the basis of what Litaor termed the "historic data set" (Litaor et al. 1995), but this component of our database was taken directly from Ripple et al. (1994). The assembled database from which our 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.

We have adjusted the observed plutonium concentrations using a generic profile of Webb et al. (1997). This profile (which, for convenience, we refer to as the "Webb profile") is based on plutonium sampling at Rocky Flats during the early 1990s. The purpose of the adjustment is to estimate the concentration in the 0–3-cm layer that would correspond to a reported concentration (Bq kg^{-1}) in a sample taken to an arbitrary depth z_d (cm). The underlying assumption is that the concentration profile in the soil column would be proportional to the Webb profile.

The Webb profile expresses the ratio of the plutonium concentration at an arbitrary depth z (cm) to the concentration averaged over the 0–3-cm layer as an empirical function of z:

$$[\mathrm{Pu}]_z = [\mathrm{Pu}]_{0-3\,\mathrm{cm}} \times [1-(1-1.41e^{-0.71z}-0.16e^{-0.192})^4] = [\mathrm{Pu}]_{0-3\,\mathrm{cm}} \cdot f(z) \qquad (4\text{-}1)$$

where the bracketed quantities represent concentrations of $^{239+240}$Pu (Bq kg^{-1}). If we are given a plutonium activity $A(z_d)$ (Bq) that was sampled from the 0–z_d-cm layer, we may express it as

$$A(z_d) = \int_0^{z_d} [\mathrm{Pu}]_z\, \rho(z)\, dz = [\mathrm{Pu}]_{0-3\,\mathrm{cm}} \int_0^{z_d} f(z)\rho(z)\, dz\,. \qquad (4\text{-}2)$$

where $\rho(z)$ (kg cm^{-3}) is the bulk density of the soil at depth z. We may solve Equation (4-2) for the desired concentration in the 0–3-cm layer:

$$[\mathrm{Pu}]_{0-3\,\mathrm{cm}} = \frac{A(z_d)}{\int_0^{z_d} f(z)\rho(z)\, dz}\,. \qquad (4\text{-}3)$$

Webb et al. (1997) provided a profile of the Rocky Flats soil bulk density, excluding rocks of diameter greater than 2 mm:

$$\rho(z) = 0.79\times10^{-3}\, z^{0.24} \quad \mathrm{kg\,cm^{-3}}\,. \qquad (4\text{-}4)$$

The exclusion of larger particles corresponds to sieving of the sample soil, which is part of the contemporary sampling protocol. Thus the density profile is not representative of the true bulk density of the soil, but on the assumption that little plutonium would be associated with rocks, it places the samples on a common basis (Webb et al. 1997). When Equation (4-4) is substituted into Equation (4-3), the integral is computed by a numerical method.

The procedure summarized by Equations (4-1) through (4-4) is directly applicable to recent samples. A strict interpretation would question its application to data taken around 1970, such as the historic subset of our database, given the temporal migration of plutonium indicated by Figure 4-1. The figure suggests that during the years 1969–1971, 80% or more of the observed

plutonium in the historic database (item 2 listed above) would have been in the 0–3-cm surface soil layer. Thus, given the evolution indicated by the figure, it is reasonable to assume that samples to a depth of 5 cm or more would ordinarily have accounted for essentially all of the 903-Area plutonium in the soil column.

We initially adjusted all concentrations in the database by the scheme given by Equations (4-1) through (4-4), irrespective of the age of the samples. We have considered readjusting the values for sample age on the basis of considerations outlined in the preceding paragraph. The result would be to decrease some of the estimated levels in the 0–3-cm layer for the 1990s. The magnitude of the change would be greatest for the shallowest sampling depths (a factor of at most 2.6 for 1 cm depths sampled in 1969) and there would be no change for 20-cm sampling depths. In view of this relatively small discrepancy and the good agreement between trends indicated by the database and by the data of Webb et al. (1997), we have not attempted an adjustment for the age of the sample. The evidence for the initial rate of decrease indicated in Figure 4-1 is tenuous, and the decline could well have been more rapid, indicating adjustments of lesser magnitude. Other considerations argued against making such an adjustment. A proper analysis of this question would require a model that would account for the evolution of the profile from the 1960s to the early 1990s when the sampling reported by Webb et al. (1997) was carried out. But it is not clear that sufficient profile data exist to support firm conclusions based on such a model.

Figure 4-2 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}. 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 of investigators to areas east of the 903 pad.

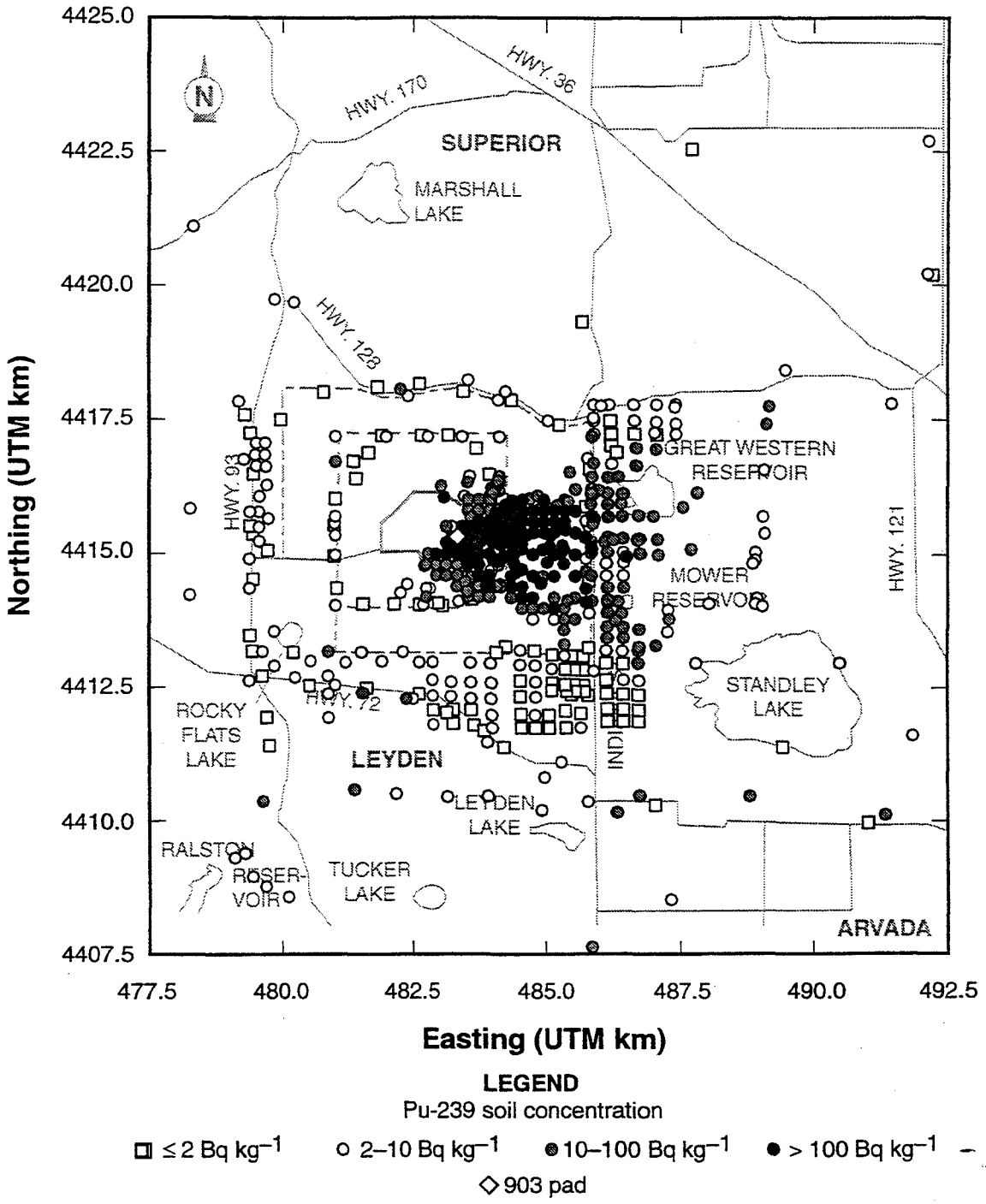

Figure 4-2. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model. The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), Ripple et al. (1994), and one series from an archive of M. Iggy Litaor provided by the Colorado Department of Public Health and Environment (CDPHE).

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To be useful, a spatial model of the plutonium concentration in soil must provide estimates for locations not included in the database (interpolation). Also, given the considerable random scatter in the data, it must provide smoothing. Some efforts have based estimation of contours on kriging methods (Litaor et al. 1995). We have based our approach 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, and thus a polar representation from this center is reasonable. Webb et al. (1997) point out that power functions[c] have given satisfactory fits to data along transects from the 903 pad. Figure 4-3 shows power functions fitted to subsets of our data base that lie near the 60°, 90°, and 120° transects; the black squares represent the data of Webb et al. (1997) (which we include in our model's data base). The data of Webb et al. (1997) are extensively documented. They provide a check on the transformation of the remaining data from heterogeneous sampling efforts to the common basis represented by the profile of Equation (4-1) and by the adjusted bulk density profile for soil particles of diameter less than 2 mm [Equation (4-4)]. The 2-mm cutoff corresponds to the sieving separation of rocks from soil used in most of the sample preparations. In some of the older samples, however, the rocks were pulverized and re-mixed with the soil (Krey et al. 1976).

The model is defined by power functions fitted to the data within each sector of 22.5°, with centerlines at 0°, 22.5°, 45°, etc. For points on a sector centerline, the model uses the value of the power function from near the 903 pad 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 pad (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. Figure 4-3 shows the data and the power function fits for the 60°, 90°, and 120° transects and indicates good consistency of the larger database with the data of Webb et al. (1997). But Figure 4-3 also emphasizes the scatter of the data, generally to a factor of about 10.

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

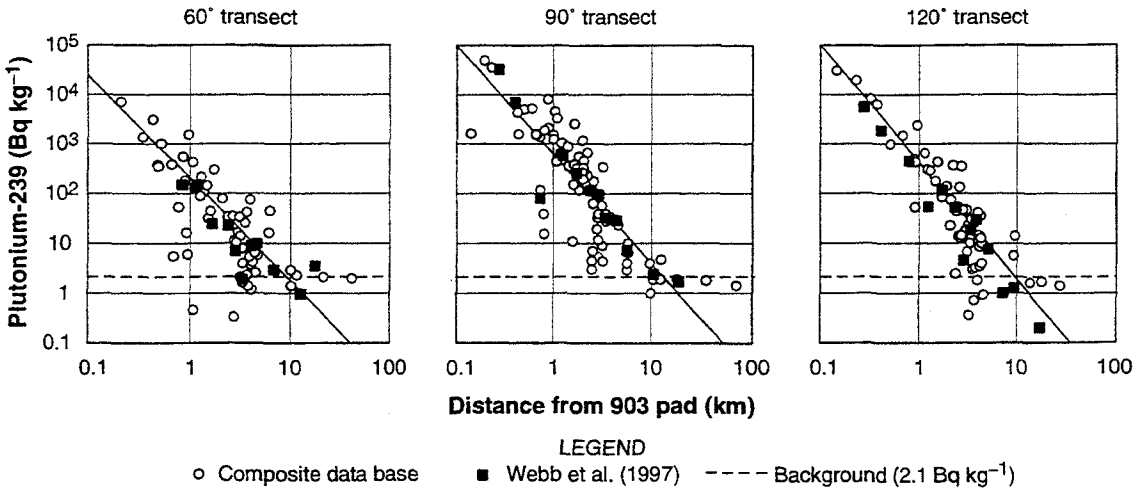

Figure 4-3. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 pad. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols.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

Contours based on the model are shown in Figure 4-4. Dashed lines indicate extrapolation of the two northwest sectors. Sample locations are shown outside the 2 Bq kg^{-1} contour (approximately background) and within the northwest sectors (where they tend to confirm the adequacy of the extrapolations). For purposes of legibility, sample points have been deleted from other regions within the contours. The contours may be considered crude, with an angular resolution no better than the linear interpolation between sectors. But they amply illustrate the considerable variation of the concentrations and the particularly rapid increase as the 903 pad is approached along eastward transects.

The model estimates are constrained not to exceed the maximum adjusted sample value (567,000 Bq kg^{-1}), which occurs in the immediate vicinity of the 903 pad. The points shown outside the 2 Bq kg^{-1} contour indicate some observations that exceed background in the 2–10 and 10–100 Bq kg^{-1} ranges. Incidents such as the 1957 fire could account for components of plutonium concentration that do not conform to the radial model. In any case, one cannot assume that these contours (or any set of contours based on plutonium concentrations in soil at Rocky Flats) provide exact partitions according to magnitude. The smoothing and interpolation provided by the model must be kept in mind. Also, the historical dose reconstruction (Rood and Grogan, 1999) predicted that releases from the 1957 fire would have progressed in a southerly direction from the plant, and deposition from this event may well have introduced perturbations which a model based on long-term wind-driven releases from the 903 Area would not predict. Elevated off-site readings near the junction of Indiana Street and Highway 72 were noted by Litaor et al. (1995), who did not speculate about the source. The model is not expected to give accurate estimates at specific locations, but rather to provide a basis for integration of resuspension fluxes over large areas for purposes of calibration, and to provide generic estimates of soil concentrations for exposure scenarios. Figure 4-3 gives a sense of the local reliability that may be expected.

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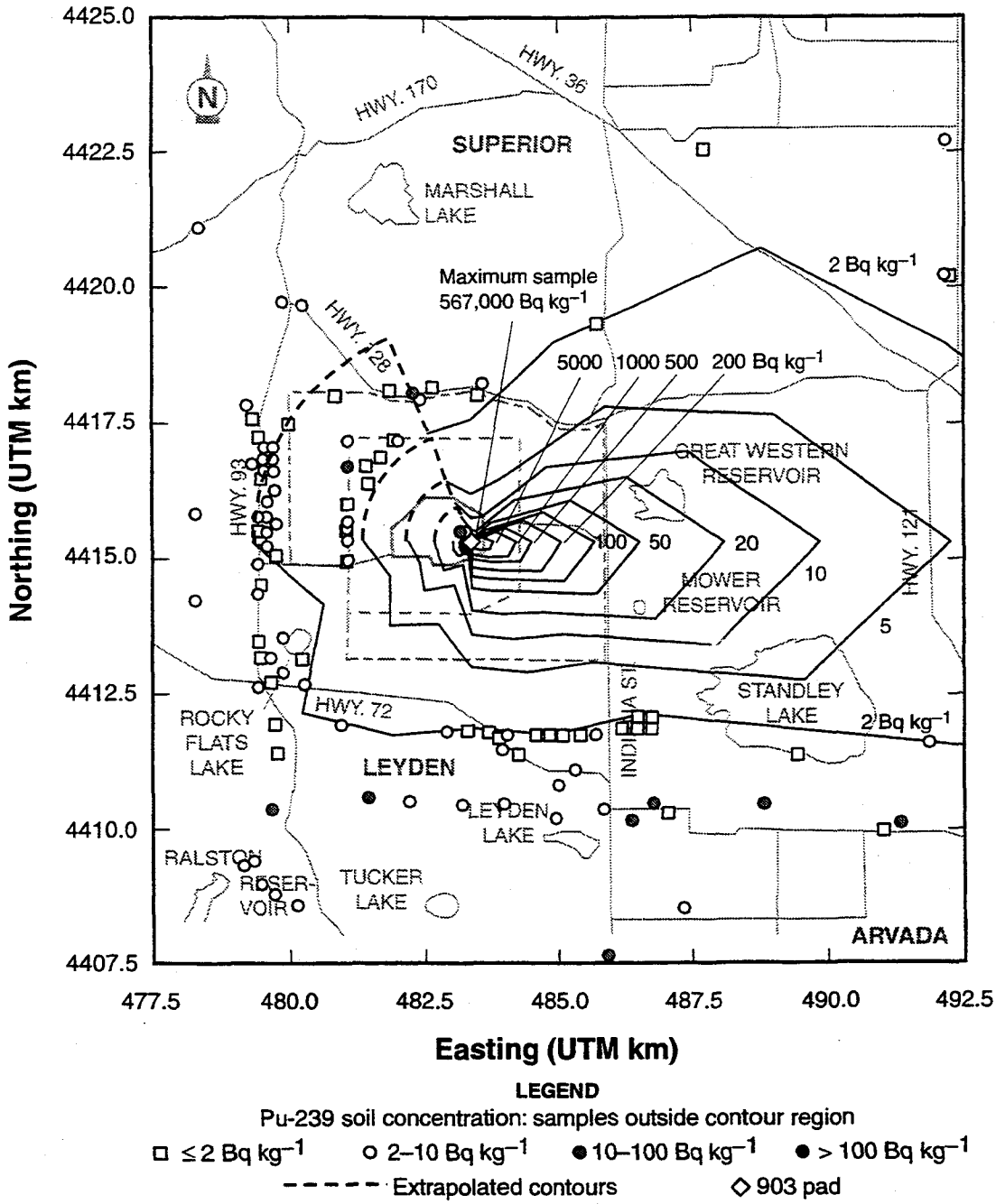

Figure 4-4. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text. Dashed parts of the contours indicate extrapolation where coverage was insufficient for fitting power functions. In these regions and outside the 2 Bq kg^{-1} 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}). The model provides interpolation and smoothing for the many measurements in the data base. It does not accurately predict concentrations at individual locations or show fine detail but is used for integration of resuspension fluxes over large areas.

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5. ESTIMATING CONCENTRATIONS OF PLUTONIUM IN AIR

The potential for resuspension of radiologically contaminated soil must be given serious consideration in assessments of the Rocky Flats site. Inhalation of airborne soil particles contaminated with plutonium and americium isotopes has been considered a possibly important exposure pathway, and its importance could be increased by a random event, such as a fire that destroyed vegetation on the site.

For the existing ground cover on most of the site (uncut grass), our simulations suggest that exposure by inhalation of resuspended soil is of less importance than the aggregate of ingestion pathways. We have replaced the internal dose coefficients in the RESRAD database, which were taken from ICRP Publication 30 (ICRP 1979), with dose coefficients currently recommended by the ICRP for members of the public (ICRP 1996). For the radionuclides of concern in the soil at Rocky Flats, the newer dose coefficients reduce the dose by inhalation and increase the dose by ingestion, and the resultant changes are substantial.

Departure from the methodology of RESRAD version 5.61 also contributes to a diminution of the inhalation pathway. The RESRAD resuspension model in version 5.61 combined a soil mass loading parameter (essentially a mass concentration of soil particles in air, g m^{-3}) with an area factor that represented the proportion of the airborne soil particles that were resuspended within the contaminated area of the soil. If the contaminated airborne particles are assumed to be representative of nearby contaminated soil, then the airborne radioactivity (Bq m^{-3}) can be calculated as the product of the soil concentration (Bq g^{-1}) and the mass loading factor (g m^{-3}). The RESRAD area factor corrects for the uncontaminated proportion of the airborne particles. The area factor in RESRAD version 5.61 was a relatively crude formulation based on a simple box model with first-order transfers of soil particles. The effect was likely to be an overestimate of the airborne radioactivity that could be as high as an order of magnitude for the uniformly contaminated sources that the model's design envisioned.

The developers of Version 5.82 of RESRAD have refined the estimate of the area factor with more realistic but somewhat more complicated assumptions (Chang et al. 1998). In our calculations, we avoided the RESRAD resuspension model altogether, replacing it by a model that takes into account the spatial variation in the soil concentration of radioactivity. The model also considers data from air monitoring in the early 1990s. This approach has the advantage of calibrating the predictions to recent site-specific data. It requires the model predictions to agree as well as possible (in a least-squares sense) with a set of observations that are believed to be representative of contemporary conditions. And it provides some quantification of uncertainty in the predictions, given the assumptions about ground cover, meteorology, and spatial distribution of plutonium in the soil. To analyze resuspension after a fire, however, the approach has to be modified, because the resuspension fluxes that are estimated from contemporary air monitoring data are influenced by the existing grass cover.

The following subsections give an overview of the models that support the inhalation pathway. Table 5-1 provides a summary of the parameters and results of resuspension fluxes that are estimated. It contains quantities that have not yet been introduced, but the reader may find it useful for backward reference during the reading of Sections 5.1–5.3.

Table 5-1. Summary of Parameters and Numeric Results for Resuspension Fluxes

Quantity	Estimate	Reference
Friction velocity u_*, Rocky Flats annual average (m s^{-1})		
Uncut grass ($z_0 = 0.05$ m)	0.231	Computed
Unvegetated soil ($z_0 = 0.01$ m)	0.211	Computed
Resuspension flux parameters		
Nevada Test site		
F_0 (mg m^{-2} s^{-1})	0.732	Anspaugh et al. 1975
γ (dimensionless)	2.09	Anspaugh et al. 1975
Flux $F = F_0 u_*^{1+\gamma}$ at 1 m (mg m^{-2} s^{-1})	6.0×10^{-3}	Computed
Rocky Flats — uncut grass		
Bootstrap-estimated flux		
Geometric mean: F (mg m^{-2} s^{-1})		
Ground-level, raw unadjusted for sampler efficiency	5.38×10^{-5}	Computed
1-m height, adjusted for sampler efficiency	2.76×10^{-5}	Computed
Geometric standard deviations		
Long-term trend	1.16	Computed
Short-term variability	3.03	Computed
Total (effective)	3.06	Computed
Adjusted total for conservatism	4	Assumed
Rocky Flats — fire scenario		
Range of resuspension factors (m^{-1})	$10^{-9} - 10^{-5}$	Sehmel 1984
Conversion parameters		
Thickness of soil layer available for resuspension (m)	0.001	Assumed
Soil bulk density (g cm^{-3})	1.3	NCRP 1999
Corresponding resuspension flux range (mg m^{-2} s^{-1})	3.3×10^{-5} – 0.33	Computed
Logarithmic midpoint of flux range (mg m^{-2} s^{-1})	3.3×10^{-3}	Computed
Geometric standard deviation	16	Computed

5.1 Model of Resuspension and Atmospheric Transport

We assume that resuspension from the Rocky Flats site is predominantly wind-driven and passively moderated by the soil, topography, and ground cover. The view taken is of a steady-state condition of the ground and source and the effect of annually averaged winds. This assumption does not deny the existence of frequent transient contributions from vehicular traffic and human or wildlife activities that disturb the soil. Indeed, an important contributor to the spatial distribution of plutonium was the grading of the 903 pad in preparation for the paving surface that was laid down in 1969. But the effect of wind seems likely to be dominant for the site as a whole and for the present and future times envisioned by the scenarios. The resolution of this assessment is insufficient to consider other more localized agents.

An empirical power function model of wind-driven material flux in a column of air, attributed to D.A. Gillette and J.H. Shinn, is

$$F = F_0 (u_* / u_0)^{1+\gamma} \tag{5-1}$$

(Anspaugh et al. 1975) where F is the predicted material flux (mg m^{-2} s^{-1}) at a reference height (say 1 m), F_0 (mg m^{-2} s^{-1}) is a flux coefficient, u_* (m s^{-1}) is the friction velocity, and u_0 is the unit friction velocity 1 m s^{-1} (so that the parenthesized ratio is dimensionless). The exponent term γ and the coefficient F_0 are usually determined from joint observations of F and u_*. The friction velocity u_* depends on the roughness of the surface, the wind speed, and the atmospheric stability. The friction velocity varies little with the height at which the wind speed is measured. Note that in this interpretation, the model predicts the vertical flux of soil mass, which would not be affected by spatial or temporal variations of radioactivity concentration in the surface soil. The mass flux would naturally be expected to vary from one site location to another, but we consider the estimate an average over the site and (as we discuss in Section 5.2) over time.

For site conditions that have existed in the 1990s, we estimate the parameters F_0 and γ by a regression procedure that depends on the spatial distribution of plutonium in the soil on and near the site (Section 5) and on air monitoring data for plutonium, measured at on-site and peripherally-located samplers during the years 1992 through 1994. The regression requires the plutonium air concentrations predicted by the resuspension flux F, used with an atmospheric transport model, to agree as well as possible (in a least-squares sense) with the observations. The regression results may thus be viewed as a calibration of the model to site-specific data. The distribution of residuals (differences of predictions and observations) estimated by the regression provides a component of uncertainty for the predictions. We discuss these matters further in Section 5.2.

The role of the ground cover at the site is not explicit in the model. In field studies that refer to the model of Equation (5-1), the soil is usually bare or sparsely vegetated (Anspaugh et al. 1975). Langer (1991) makes a case for research that would support explicit representation of the grass in modeling deposition and resuspension. Such modeling might account for transfer of radioactivity from soil to grass leaves by rain splash and the role of leaf motion in the wind in reentraining the radioactivity into the air stream. Our interpretation, however, considers the aggregate effect of such mechanisms to be in steady state at the scale of one year, and this aggregate effect is assumed implicit in the parameters of Equation (5-1). With this understanding, we apply the model to the Rocky Flats site with its contemporary grass cover.

The predictions of the model of Equation (5-1) with the data from two locations cited by Anspaugh et al. (1975) exceed those of the same model calibrated to the Rocky Flats site under contemporary conditions with the existing ground cover. In one case, based on data from the Nevada Test Site, the estimated resuspension flux (6.0×10^{-3} mg m^{-2} s^{-1}) exceeds our estimate for Rocky Flats (2.76×10^{-5} mg m^{-2} s^{-1}) by a factor of more than 200 (an annual average of the friction velocity u_* based on Rocky Flats meteorological data was used for the comparison). But a range of other observations reviewed by Sehmel (1984) suggests that even higher material fluxes might prudently be included in uncertainty estimates for Rocky Flats for the case of a fire. Section 5.3 gives details of such a fire scenario.

5.2 Nonlinear Regression Based on Air-Monitoring Data to Estimate Resuspension Parameters

Plutonium from the 903 pad has been unevenly dispersed over the site, and the soil particles to which it is attached continue to be resuspended, making the plutonium available for inhalation.

Using the interpolation model of soil contamination described in Section 5 with the soil flux model of Equation (5-1) and an atmospheric transport model, we are able to estimate the concentration of the resuspended plutonium in the air at any specified location, given the values of F_0 and γ. This approach was contemplated by Anspaugh et al. (1975), who suggested that "the vertical flux of contaminant may be predicted by the dust flux calculated using Equation (5-1) multiplied by the amount of contaminant per unit mass of soil surface material. This information may then be combined with a suitable model of atmospheric transport and diffusion which also considers the areal distribution of the contaminant to calculate airborne concentrations of resuspended contaminant both within and outside the contaminated area." We now summarize this model.

5.2.1 The atmospheric transport and diffusion model

The model of transport and diffusion used for our calculations is the Gaussian plume for a point source (Barr and Clements 1984). The Gaussian plume prediction for each combination of source and receptor points is multiplied by the source strength (i.e., the resuspension flux at ground level in Bq m^{-2} s^{-1}) and by the differential area (m^2) and integrated over the contaminated region. The process is repeated for each of 16 wind directions, 6 wind speeds, and 6 atmospheric stability categories, and the results are averaged with weights from the meteorological data for the site. The weighted average is an estimate of the $^{239+240}$Pu concentration in the air at the specified location. The procedure is summarized by the equation

$$\chi_{ijk}(x,y,z) = F_{ik} \iint_R C_s(\xi,\eta)\, G_{ijk}(x,y,z;\xi,\eta,z_0)\, d\xi\, d\eta \qquad (5\text{-}2)$$

where $\chi_{ijk}(x,y,z)$ is the predicted annual average air concentration at easting and northing coordinates (x,y) and height z above the ground. The subscripts ijk correspond, respectively, to wind speed (6 discrete categories), wind direction (16 sectors), and atmospheric stability (6 categories A–F). The subscripts ik on the resuspension flux F reflect the dependence of the flux on the friction velocity u_*, which depends on wind speed and atmospheric stability. The $^{239+240}$Pu concentration in soil at the source point with easting and northing coordinates (ξ,η) is $C_s(\xi,\eta)$ (Bq kg^{-1}). The symbol $G_{ijk}(x,y,z;\xi,\eta,z_0)$ denotes the Gaussian plume prediction of $^{239+240}$Pu (Bq m^{-3}) at (x,y,z) corresponding to a unit flux (1 Bq m^{-2} s^{-1}) at source location (ξ,η) and height z_0. (The parameter z_0 is the *roughness height* for the terrain and corresponds to the height above ground where the horizontal wind speed becomes zero. We use this height for a ground-level release, i.e., resuspension.)

The double integral ($\iint$) in the formula of Equation (5-2) represents an addition of all differential point-source contributions throughout the contaminated region R, taking into account the varying level of concentration in the soil and the location of the source relative to the receptor. Contamination at background level from a wider area is also included, when it is appropriate to do so (for example, in the regression procedure described in Section 5.2.2). If the wind direction is from the receptor toward the source point, the result is zero. If the wind is blowing from the source directly toward the receptor point, the receptor concentration will vary with the source concentration and the distance downwind.

Equation (5-2) represents the concentration at receptor location (x, y) and height z for a single wind speed, wind direction, and atmospheric stability category, specified by the subscripts ijk. A meteorological joint frequency table (JFT) for the Rocky Flats site, based on annual averages of observations, provides a relative frequency for each combination of these three factors, denoted by w_{ijk}. The annual average concentration of $^{239+240}$Pu in air at the receptor location is given by

$$\overline{\chi}(x, y, z) = \sum_{i,j,k} w_{ijk} \chi_{ijk}(x, y, z) \tag{5-3}$$

where we are assuming that the weights are normalized so that the sum of w_{ijk} over all i, j, and k is 1.

The transport and diffusion model includes provision for modifying the soil concentration levels of plutonium so that for any specified value C_0, the all points interior to the contour line $C = C_0$ have concentration C_0 rather than the concentrations indicated by Figure 4-4. This truncation arrangement facilitates calculations that simulate different levels of remediation, whereas the model without this feature represents the site without remediation (other than earlier cleanup work that might be reflected in the database). The feature is useful for explicit calculations to compare with RSALs computed by methods that do not simulate remediation. We describe such calculations in Section 7.

The point-source Gaussian plume model G_{ijk} used in our calculations was adapted primarily from the formulation used for the Environmental Protection Agency's ISC3 (ISC = Industrial Source Complex) model for atmospheric advection and diffusion (EPA 1995). However, to account for material loss from the plume due to deposition, we used a source-depletion model similar to the scheme of Van der Hoven (1968) rather than the somewhat more complicated surface-depletion representation used in ISC3 (short-term). The latter method is considered more realistic, but the extra effort needed to implement it and the possible increase in computer running time argued against it for this application.

Note that the flux F in Equation (5-2) depends on the parameters F_0 and γ. The regression procedure for determining these parameters is discussed in Section 5.2.2.

5.2.2 Nonlinear regression for F_0 and γ using on-site and peripheral air monitoring data

We have chosen to base the regression on the annual data from 34 samplers of the S-series for the years 1992, 1993, and 1994. These data provide spatial and some temporal information about variation of the air concentrations of $^{239+240}$Pu resuspended from the soil on and near the site. We restricted attention to these samplers because of a common protocol and their extensive (although not comprehensive) coverage of the contaminated area. Beginning with the year 1992, the annual monitoring reports tabulated sufficient precision for the sampler data to permit us to distinguish among concentrations of similar magnitude. Previously, for example, 0.0001 might have been recorded for any value greater than or equal to 0.00005 and less than 0.00015 (a factor of three). Obtaining original data sheets might have permitted us to extend the record to take in additional years and samplers, but considerable data analysis would have been required for the numerous adjustments necessary to put all of the data on a credibly common basis. Moreover,

there is no assurance that a coherent picture would have emerged. Corrections would have to account for different sampler characteristics and the longer-term temporal trend in the soil concentrations of $^{239+240}$Pu.

We use an index m to identify the sampler location, and we write $\overline{\chi}(m \mid F_0, \gamma)$ for the predicted annual concentration at sampler m, given the values of the parameters F_0 and γ. This predicted value includes the estimated contribution from background-level plutonium (fallout) contamination carried by soil within and beyond the region contaminated by Rocky Flats operations. For the observed annual average value at sampler m for year t (= 1992, 1993, or 1994), we write $\overline{\chi}_{mt}$. (The predictive model is not a function of the year t.) The least-squares problem associated with the regression is to find F_0 and γ such that

$$S = \sum_{m,t} [\ln \overline{\chi}_{mt} - \ln \overline{\chi}(m \mid F_0, \gamma)]^2 = \text{minimum}, \tag{5-4}$$

where the summation is taken over $m = 1, \ldots, 18$ samplers and $t = 1992, \ldots, 1994$. This problem is nonlinear in γ and linear in F_0.

The logarithmic residuals $r_{mt} = \ln \overline{\chi}_{mt} - \overline{\chi}(m \mid \hat{F}_0, \hat{\gamma})$ corresponding to the solution $F_0 = \hat{F}_0$ and $\gamma = \hat{\gamma}$ of Equation (5-4) are used to estimate a geometric standard deviation for the predicted air concentration:

$$\text{GSD}_{\text{resid}} = \exp\left(\left[\frac{1}{M-2} \sum_{m,t} r_{mt}^2 \right]^{1/2} \right) \tag{5-5}$$

where $M = 34 \times 3 = 102$ observations (two degrees of freedom are subtracted for the estimation of the two parameters F_0 and γ). This geometric standard deviation represents components of uncertainty associated with spatial variability (samplers in different locations) and temporal variability (annual averages for different years). The assumptions underlying the regression treat the observations as if they include error distributions that are identical and independent from location to location and from year to year. Table 5-2 shows the sampler locations, the observed annual average air concentrations of $^{239+240}$Pu for 1992–1994, and the predictions based on the fitted model. Table 5-3 gives parameter estimates from the regression.

The regression estimates the exponent of the flux model in Equation (5-1) as $1 + \hat{\gamma} = 1.44$ and the reference flux as $\hat{F}_0 = 3.62 \times 10^{-4}$ mg m^{-2} s^{-1}. The residual geometric standard deviation estimated by the regression (Equation 5-5) is $\text{GSD}_{\text{resid}} = 3.03$, which, for a lognormal distribution, corresponds to a 95th to 50th percentile ratio of $3.03^{1.65} \approx 6.2$. Figure 5-1 shows the predicted $^{239+240}$Pu air concentrations plotted against the observations for the S-series samplers at Rocky Flats.

Table 5-2. Sampling Locations with Predicted and Observed Plutonium Air Concentrations

Station	Distance (km)	Angle (°)	Plutonium-239+240 in air (Bq m^{-3}) Predicted	Observed 1992	Observed 1993	Observed 1994
S-03	0.996	321.0	1.764×10^{-7}	1.590×10^{-7}	1.230×10^{-7}	7.030×10^{-8}
S-04	0.760	348.9	2.971×10^{-7}	2.340×10^{-7}	4.740×10^{-7}	6.070×10^{-7}
S-05	0.575	357.2	3.418×10^{-7}	9.930×10^{-7}	1.320×10^{-6}	2.500×10^{-6}
S-06	0.356	30.6	1.470×10^{-6}	9.230×10^{-6}	2.890×10^{-6}	1.970×10^{-6}
S-07	0.164	77.7	2.144×10^{-5}	7.170×10^{-6}	6.020×10^{-6}	7.252×10^{-6}
S-08	0.159	105.3	3.968×10^{-5}	1.850×10^{-5}	1.310×10^{-5}	1.490×10^{-5}
S-09	0.163	140.2	1.106×10^{-5}	2.040×10^{-5}	6.340×10^{-6}	1.130×10^{-5}
S-10	0.340	222.5	4.243×10^{-7}	2.690×10^{-7}	1.820×10^{-7}	1.720×10^{-6}
S-11	0.713	243.9	1.837×10^{-7}	2.430×10^{-7}	4.710×10^{-7}	2.810×10^{-7}
S-13	1.140	258.0	6.195×10^{-8}	9.880×10^{-8}	1.550×10^{-7}	1.350×10^{-7}
S-14	1.566	265.4	7.598×10^{-8}	3.070×10^{-8}	9.770×10^{-8}	4.920×10^{-8}
S-16	1.265	297.6	8.752×10^{-8}	6.480×10^{-8}	1.650×10^{-7}	6.180×10^{-8}
S-17	0.594	274.7	2.225×10^{-7}	3.150×10^{-7}	4.550×10^{-8}	1.390×10^{-7}
S-18	0.154	264.8	1.184×10^{-6}	6.570×10^{-7}	5.300×10^{-7}	5.910×10^{-7}
S-19	0.084	311.6	1.971×10^{-6}	1.220×10^{-6}	8.880×10^{-7}	5.070×10^{-7}
S-20	0.392	347.7	8.824×10^{-7}	6.360×10^{-7}	7.770×10^{-7}	2.890×10^{-7}
S-21	0.864	20.8	2.377×10^{-7}	2.340×10^{-7}	2.930×10^{-7}	2.960×10^{-7}
S-22	1.167	34.5	2.242×10^{-7}	2.380×10^{-7}	3.080×10^{-7}	2.780×10^{-7}
S-23	0.660	225.0	1.587×10^{-7}	1.310×10^{-7}	1.390×10^{-7}	6.290×10^{-8}
S-24	1.636	46.7	2.949×10^{-7}	1.340×10^{-7}	1.080×10^{-7}	5.250×10^{-8}
S-25	0.592	334.9	2.563×10^{-7}	2.775×10^{-6}	7.950×10^{-6}	1.210×10^{-6}
S-31	3.816	264.2	1.869×10^{-8}	2.700×10^{-8}	2.520×10^{-7}	1.630×10^{-8}
S-32	4.069	291.6	2.736×10^{-8}	1.670×10^{-8}	2.480×10^{-8}	3.700×10^{-8}
S-33	4.749	314.3	2.766×10^{-8}	2.590×10^{-8}	3.700×10^{-8}	3.330×10^{-8}
S-34	2.679	15.8	3.757×10^{-8}	2.700×10^{-8}	5.550×10^{-8}	1.850×10^{-8}
S-35	3.171	47.8	1.405×10^{-7}	3.700×10^{-8}	2.700×10^{-8}	2.330×10^{-8}
S-36	2.697	61.4	8.688×10^{-8}	4.030×10^{-8}	4.700×10^{-8}	2.890×10^{-8}
S-37	2.361	85.8	3.409×10^{-7}	8.730×10^{-8}	7.400×10^{-8}	8.330×10^{-8}
S-38	2.496	108.4	4.262×10^{-7}	6.070×10^{-8}	5.100×10^{-7}	7.400×10^{-8}
S-39	3.672	139.5	4.439×10^{-8}	3.370×10^{-8}	4.000×10^{-8}	3.700×10^{-8}
S-40	4.515	149.1	2.756×10^{-8}	2.530×10^{-7}	4.000×10^{-8}	3.700×10^{-8}
S-41	2.874	189.3	2.467×10^{-8}	3.400×10^{-8}	4.630×10^{-8}	3.290×10^{-8}
S-42	4.019	230.6	1.220×10^{-8}	6.180×10^{-8}	1.390×10^{-7}	1.370×10^{-8}
S-43	4.839	237.5	1.100×10^{-8}	3.590×10^{-8}	3.700×10^{-8}	4.510×10^{-7}

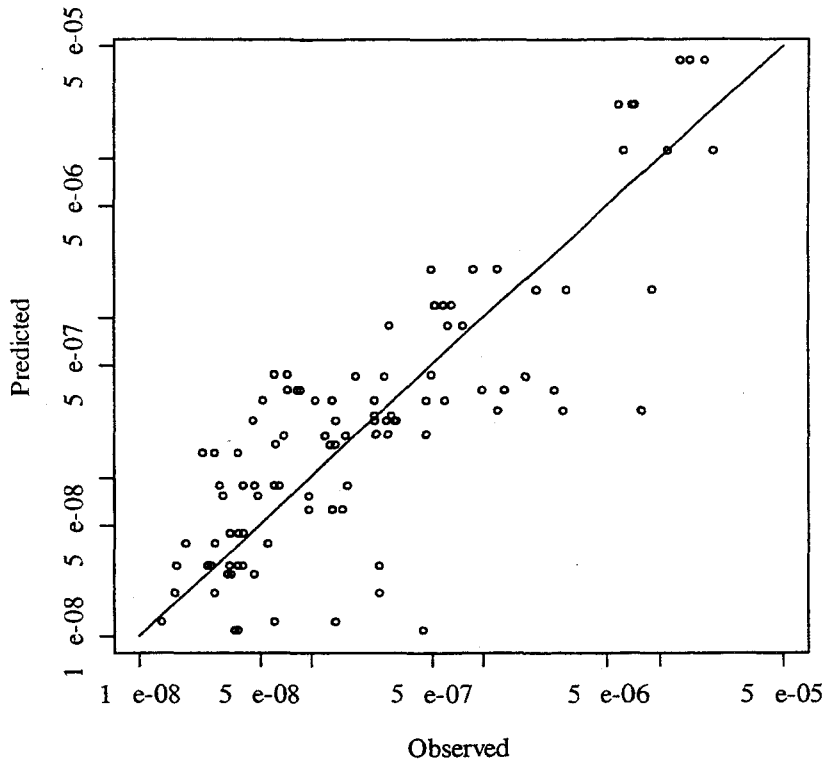

Figure 5-1. Model-predicted plutonium concentrations (Bq m^{-3}) plotted against observations of $^{239+240}Pu$ at the Rocky Flats S-series samplers. The units for both axes are Bq m^{-3}. The line is the locus of points for which the prediction equals the observed value.

Table 5-3. Regression Parameter Estimates for Resuspension Flux

Parameter	Mean	Standard deviation
Exponent ($\gamma+1$, dimensionless)	1.44	0.588
Reference flux (F_0, mg m^{-2} s^{-1})	3.62×10^{-4}	2.92×10^{-4}
Standard deviation of logarithmic residuals: 1.11		
Geometric standard deviation: 3.03		
Degrees of freedom: 100		
Correlation coefficient for $\gamma+1$ and $'F_0$: 0.992		

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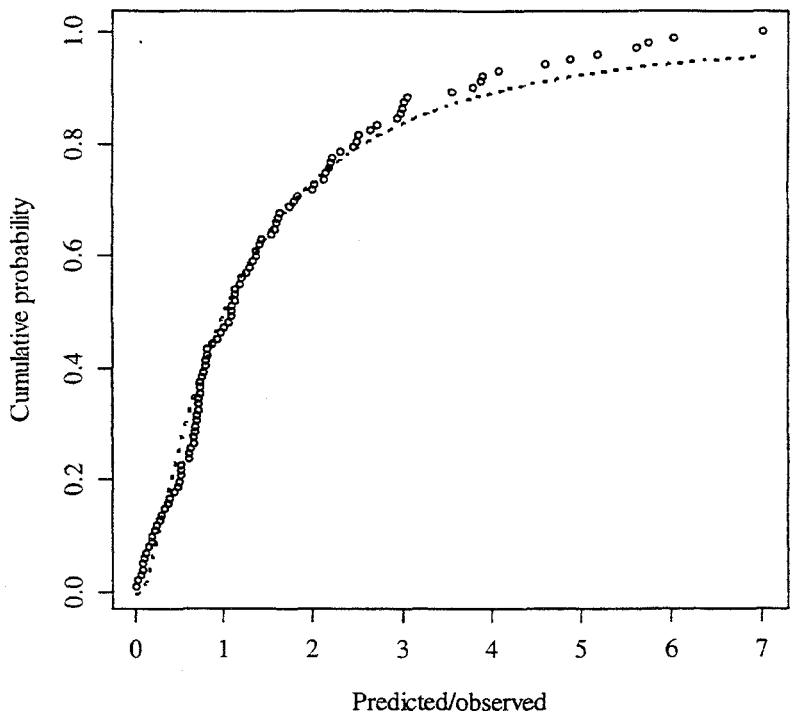

Figure 5-2. Cumulative probability plot of predicted-to-observed ratios from the regression on the sampler data. The points represent the empirical distribution function of the P/O ratios based on the logarithmic residuals. The dashed curve is the lognormal distribution with geometric mean 1.00 and geometric standard deviation 3.02 (the parameters were estimated from the empirical distribution). This lognormal distribution was used as the major component of an uncertainty factor for airborne $^{239+240}$Pu resuspended from the Rocky Flats site.

The regression summarized in Table 5-3 does not provide detailed information on the joint uncertainty distribution of F_0 and γ, or more to the point, the distribution of the estimated flux $F = F_0 u_*^{1+\gamma}$, from which air concentrations of resuspended dust and attached particles of $^{239+240}$Pu can be estimated for locations other than those of the air samplers. If the regression were linear in both parameters, standard assumptions and theory would indicate a bivariate normal distribution for the pair, but the marginal (individual) distributions are likely skewed. An approach that provides a quantitative estimate of the joint distribution is the bootstrap (Efron 1982; Efron and Tibshirani 1998). A bootstrap procedure treats the residuals of the standard regression as an empirical prototype of the distribution of errors, and one calculates distributions of the parameters by repeated Monte Carlo sampling of this distribution, with replacement.

Figure 5-3 is based on such a bootstrap estimation procedure, in which the residuals were resampled with replacement 1000 times. The result of a resampling is a randomly perturbed version of the prototype distribution of residuals. At each Monte Carlo realization, the perturbed residual distribution was used to calculate a corresponding set of "observations," and a new pair of parameters (F_0, γ) was determined from the least-squares method of Equation (5-4). This

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figure — which plots F_0 against the exponent $1+\gamma$ — indicates that the two parameter estimates are highly correlated. Their relationship is represented by a cubic polynomial that was fitted to the data, although the fitted curve plays no role in our subsequent calculations.

The empirical distribution of the resuspension flux F (Equation 5-1), which is calculated from the bootstrap data using the annual average value of the friction velocity u_*, is shown in Figure 5-4. For any location (x, y), the annual average air concentration of $^{239+240}$Pu is approximately proportional to this flux distribution:

$$\chi(x, y, z) \approx F \cdot G(x, y, z; C_{\text{soil}}). \tag{5-6}$$

The function G, which is based on the spatially integrated Gaussian plume, is a function of position and height and the spatial distribution C_{soil} of $^{239+240}$Pu in the soil.

The distribution of χ in Equation (5-6) corresponds to what might be interpreted as a theoretical long-term mean flux under steady-state conditions. But if we think of the location and year as being chosen at random, such a distribution does not account for all components of uncertainty in the estimate of plutonium concentration at that location during that year. The missing component is supplied by the distribution of the residuals r_{mt} (Equation 5-5), which represent the spatial and temporal variability of the air concentration.

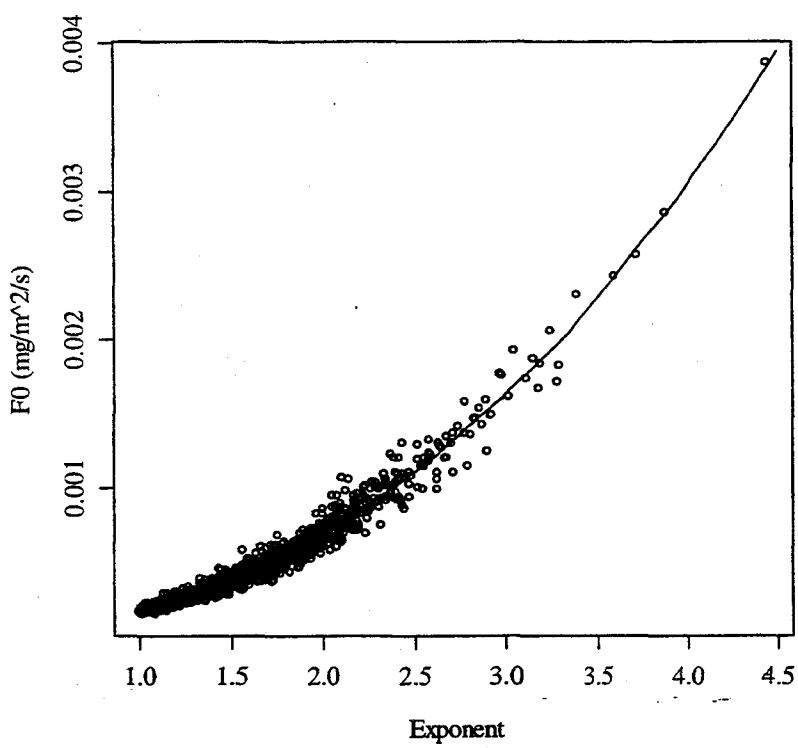

Figure 5-3. Distribution of points (Exponent = $1+\gamma, F_0$) from the bootstrap regression of predicted air concentration on the sampler data. The units of the reference flux F_0 are mg m^{-2} s^{-1}, and the exponent $1+\gamma$ is dimensionless. The fitted curve is the cubic polynomial

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$f(\theta) = c_0 + c_1\theta + c_2\theta^2 + c_3\theta^3$, with $\theta = 1+\gamma$ and coefficients $c_0 = 5.688\times10^{-5}$, $c_1 = -6.749\times10^{-5}$, $c_2 = 1.795\times10^{-4}$, $c_3 = 6.032\times10^{-6}$.

It will be useful to express the total uncertainty as a normalized factor that will be multiplied by any deterministically calculated air concentration of $^{239+240}$Pu on or near the site. We will assume that this uncertainty factor is independent of position and time, and that it expresses the two components of uncertainty described above (uncertainty in the estimate of a long-term and spatially averaged theoretical mean, and short-term local variability about that mean). Finally, we will increase the variance of this factor somewhat to compensate for a sample of convenience that may be inadequately representative of the longer term and unmonitored parts of the site. The product of two independently distributed random variables having the lognormal distribution pictured in Figure 5-2 and the one shown in Figure 5-4, respectively, is lognormally distributed with geometric mean GM = $1 \times 5.31 \times 10^{-5}$ and geometric standard deviation

$$\mathrm{GSD} = \exp\sqrt{(\ln 3.03)^2 + (\ln 1.16)^2} = 3.06 \qquad (5\text{-}7)$$

Thus, the estimated uncertainty of the long-term mean accounts for little of the variance of the composite uncertainty. This GSD corresponds to a 95th/50th-percentile ratio of about 5.3. Increasing the GSD to 4.0 gives a distribution with 95th/50th-percentile ratio of approximately 10. We make this precautionary adjustment as a measure of conservatism in the calculations.

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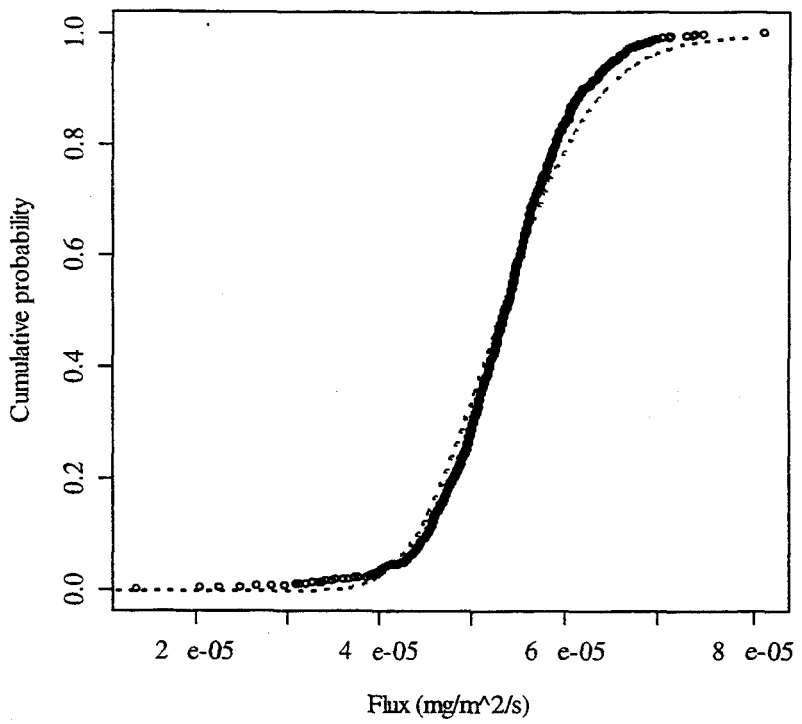

Figure 5-4. Empirical cumulative distribution function of estimated soil flux (solid curve) with fitted lognormal distribution (dashed curve). The parameters of the lognormal distribution are geometric mean: 5.31×10^{-5} mg m^{-2} s^{-1}, and geometric standard deviation: 1.16. The bootstrap median (5.38×10^{-5} mg m^{-2} s^{-1}) has been used as the reference value. This parameter is interpreted as the ground-level flux and must be adjusted for the 1-m height.

We now rewrite Equation (5-6) as

$$\chi(x, y, z) = \eta \hat{F} \cdot G(x, y, z; C_{\text{soil}}) \tag{5-8}$$

where $\hat{F} = 5.38 \times 10^{-5}$ mg m^{-2} s^{-1} is the median resuspension flux and η is the lognormal uncertainty factor just derived (GM = 1, GSD = 4).

In statistics textbooks, the kind of prediction that we have outlined is discussed in the context of prediction of a value for a new member of the population represented by the regression data. For simple linear regression, the exact formula for the standard error as a function of the independent variable can be worked out explicitly (Snedecor and Cochran 1967, Section 5.12). For multiple linear regression, Kendall and Stuart (1967) discuss confidence intervals for the expected value of the dependent variable and for a new observation, given a specified value of the regressor vector (in this case, a particular sampling station and year).

5.2.3 Adjusting the estimated flux for activity distribution and sampler efficiency

The flux estimate in the previous section (geometric mean 5.38×10^{-5} mg m^{-2} s^{-1}) is estimated as if the $^{239+240}$Pu activity were uniformly distributed with respect to the mass of the soil

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particles and the sampler efficiency were 100%. But it is physically plausible that the radioactivity would be more nearly distributed according to particle surface area. Also, the monitoring samplers would account for the plutonium with decreasing efficiency as aerodynamic diameter of the carrier particles increases. On the basis of assumptions given below, we estimate that the overall monitoring sampler efficiency for $^{239+240}$Pu attached to resuspended soil particles at Rocky Flats, is about 80%. If we restrict consideration to particles of aerodynamic diameter less than 15 μm, the efficiency is 93%.

To arrive at such a conclusion, one must make some assumption about the distribution of aerodynamic diameters of the particle population. Following guidance summarized in NCRP Report No. 129 (NCRP 1999) from primary sources, we assume that the distribution of particle aerodynamic diameters is lognormal with geometric mean between 2 μm and 6 μm and geometric standard deviation 5 (distribution with respect to mass is assumed for this specification; we use 4 μm for these calculations). Larger components are sometimes seen, but these components are considered transitory and are usually associated with sandy soils. Accordingly, we neglect them for this site. We assume further that the plutonium radioactivity is distributed according to the surface areas of the particles.

We have fitted an efficiency curve to data for high-volume samplers of the type used for monitoring at Rocky Flats (Figure 5-5). These data are discussed by Rope et al. (1997). Using this efficiency curve and the foregoing assumptions, we may express the sampler efficiency as

$$E = \frac{\int_0^\infty D^2 \varepsilon(\sqrt{\rho} D) f_c(D)\, dD}{\int_0^\infty D^2 f_c(D)\, dD} \tag{5-9}$$

where the physical particle diameter D (μm) is the variable of integration. The function ε gives the sampler efficiency as a function of aerodynamic diameter $\sqrt{\rho} D$; the value $\rho = 2.6$ g cm^{-3} is a reasonable generic (physical) density for soil particles. The lognormal probability density function f_c has physical geometric mean GM_c that corresponds to the particle count. The conversion is $\mathrm{GM}_c = \mathrm{GM}_m \exp(-2 \ln^2 \mathrm{GSD})$ (Seinfeld 1986), where GM_m is the physical geometric mean corresponding to distribution with respect to mass (or more accurately, volume). The geometric standard deviation (GSD) is the same for both distributions. For particles in a restricted range, the infinite upper limits in the integrals of Equation (5-9) are replaced by the maximum physical diameter for the range. For example, for particles in the aerodynamic range of 0–15 μm, we would use the value $15/\sqrt{2.6} = 9.3\ \mu\mathrm{m}$ as the physical upper limit. As noted in the introductory paragraph, the efficiency when the total distribution is considered (infinite upper limits), the efficiency is $E = 0.80$. When attention is restricted to the aerodynamic range 0–15 μm, the efficiency is $E = 0.93$.

The adjustment of our median flux estimate for sampler efficiency is $5.38 \times 10^{-5} / 0.80 = 6.72 \times 10^{-5}$ mg m^{-2} s^{-1}. Because of the role this quantity plays in the Gaussian model, however, it should be interpreted as a flux at ground level, rather than a flux at 1 m, which is a frequent reference height for experimental measurements.

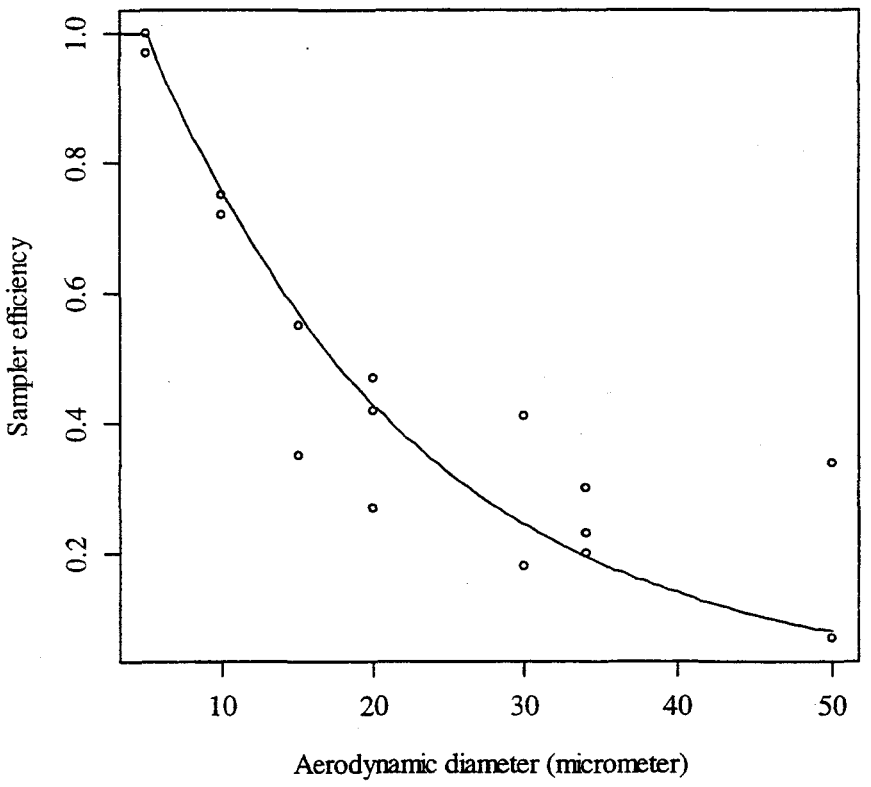

Figure 5-5. Sampler-efficiency curve fitted to data discussed by Rope et al. (1999). Some of the scatter in the plotted points is accounted for by different orientations of the device to the airflow. The function has the form $\varepsilon(D) = \exp(-b(D-5))$ ($b = 0.0563\ \mu\text{m}^{-1}$) for aerodynamic diameter $D \geq 5\ \mu\text{m}$ and $\varepsilon(D) = 1$ otherwise.

We represent ground level with the roughness height, $z_0 = 0.05$ m, the value corresponding to uncut grass. This parameter is interpreted as the height at which the horizontal wind speed profile effectively reaches zero, given the surface texture that impedes the wind flow. We make use of the fact that the mass concentration of suspended particles as a function of the height z tends to follow a power function:

$$\chi(z) = \chi(1) \cdot z^p \tag{5-10}$$

Also, the flux F at any height is proportional to the mass concentration at that height:

$$F(z) = -0.4 p u_* \chi(z) \tag{5-11}$$

(Anspaugh et al. 1975); the number 0.4 is the von Kármán constant, and u_* is the friction velocity, which may be considered essentially independent of height. Equations (5-10) and (5-11) can be combined to derive an equation for the flux at 1-m height in terms of the ground-level value at z_0:

$$F(1) = F(z_0) z_0^{-p}. \tag{5-12}$$

To use these equations to calculate the adjusted flux, it is first necessary to estimate the exponent p. Anspaugh et al. (1975) report estimates in the range $p = -0.35$ to -0.25. Using the average $p = -0.3$ and Equation (5-12), we estimate

$$F(1) = 6.72 \times 10^{-5} \times 0.05^{0.3} = 2.76 \times 10^{-5} \text{ mg m}^{-2} \text{ s}^{-1} \quad (5\text{-}13)$$

We adopt the estimate of Equation (5-13), $F(1) = 2.76 \times 10^{-5}$ mg m^{-2} s^{-1}, as generic for the site average flux with contemporary grass cover. For plutonium predictions with the Gaussian transport model, however, we must return to the original ground-level flux of 5.38×10^{-5} mg m^{-2} s^{-1}. We must also multiply the predicted airborne radioactivity by the sampler efficiency factor 1/0.93 for particles of aerodynamic diameter less than 15 µm, because the data were not adjusted for sampler efficiency. Using this procedure for the location of the *RAC* Resident Rancher scenario (located 300 m E of the center of the 903 pad), we obtain 2.80×10^{-5} Bq m^{-3}. For comparison, the 1992–1994 data for the samplers S-07, S-08, and S-09 have sample mean 1.25×10^{-5} Bq m^{-3} and sample standard deviation 5.83×10^{-6} Bq m^{-3} (both statistics were corrected for sampler efficiency). Each of these samplers is about 160 m from the center of the 903 pad, with directions ranging from E to SE (Table 5-2). This comparison is consistent with the regression's overprediction of most of the data for these three samplers (factors of 1.7 to 3.6 for 7 of the 9 observations). However, overprediction at three samplers does not indicate a bias; in general, the regression overpredicts and underpredicts equitably (Figure 5-1).

5.3 Resuspension from Unvegetated Soil after a Fire

If a fire should remove vegetation from large areas of the site, the bare soil would likely be subject to higher resuspension fluxes than those we have estimated for the uncut grass cover that grows on most of the site. However, existing data for resuspension factors span several orders of magnitude, and we have seen no information that credibly narrows this large uncertainty.

As we noted in Section 5.1, one of the data sets discussed by Anspaugh et al. (1975) predicts a resuspension flux (6.0×10^{-3} mg m^{-2} s^{-1}) for sparsely vegetated soil that exceeds our estimate for Rocky Flats with its uncut grass (2.76×10^{-5} mg m^{-2} s^{-1}) by a factor of more than 200. Other literature indicates both comparable and substantially larger fluxes.

Sehmel (1984) reviewed resuspension factors and tabulated the results of numerous studies. A report of Sehmel and Orgill (1973) presented a range of 10^{-9} to 10^{-5} m^{-1} for resuspension factors based on measurements at Rocky Flats in 1970 and early 1971. This was a time when much of the soil around the recently paved 903 pad may have been in a disturbed condition, although possibly not the best surrogate for a burned area. However, this range includes nearly all resuspension factors and ranges in Table 12-7 of Sehmel (1984). In particular, it includes a range labeled "Sandy soil with charred debris" for a 1300-m^2 source for wind speed less than 5 m s^{-1}. Provisionally, we use this range of 10^{-9} to 10^{-5} m^{-1} for resuspension factors for estimating a flux distribution. For total soil fluxes (assuming an infinite-extent uniform source of available soil particles 1 mm thick), this range translates into a range of 3.3×10^{-5} mg m^{-2} s^{-1} to 0.33 mg m^{-2} s^{-1}.

The logarithmic midpoint of this range (3.3×10^{-3} mg m^{-2} s^{-1}), which we associate with bare soil, is about 120 times the median regression-based flux (2.76×10^{-5} mg m^{-2} s^{-1}) corresponding to the existing ground cover and within a factor of two of the estimate for the Nevada Test Site (6.0×10^{-3} mg m^{-2} s^{-1}) based on parameters from Anspaugh et al. (1975).

The foregoing conversion of resuspension factors to total fluxes made use of the equation

$$R = -0.4 p u_* S_f \tag{5-14}$$

(Anspaugh et al. 1975), where R is the resuspension rate (s^{-1}), $p = -0.3$ (Section 5.2.3), and $u_* = 0.211$ m s^{-1}, an annual average value for Rocky Flats assuming bare soil. Substituting these values in Equation (5-14) gives the value $R = 2.53 \times 10^{-9}$ s^{-1}. The volume V containing the particles (which we assumed to be a layer 1 mm thick) is 10^{-3} m^3 m^{-2}, and the soil bulk density is assumed to be $\rho_b = 1.3 \times 10^9$ mg m^{-3} (1.3 g cm^{-3}). Thus $F = RV\rho_b = 2.53 \times 10^{-9}$ $s^{-1} \times 10^{-3}$ m^3 $m^{-2} \times 1.3 \times 10^9$ mg $m^{-3} = 3.3 \times 10^{-3}$ mg m^{-2} s^{-1}. This quantity is obviously sensitive to the assumed thickness of the layer of soil available for resuspension.

Using the logarithmic flux midpoint 3.3×10^{-3} mg m^{-2} s^{-1} derived above and the Gaussian atmospheric transport model, we estimated a total $^{239+240}Pu$ concentration in air of 5.44×10^{-3} Bq m^{-3} at the primary location of the Resident Rancher scenario (300 m east of the 903 pad). This estimate exceeds the median value for grass cover (2.80×10^{-5} Bq m^{-3}) by a factor of about 200. In the calculation, we increased the geometric mean of the aerodynamic diameter distribution with respect to mass to 6 µm (NCRP 1999), retaining the geometric standard deviation 5. For this particle size distribution, we estimate that 98% to 99% of the $^{239+240}Pu$ radioactivity is associated with particles of aerodynamic diameter less than 15 µm, and accordingly we make no adjustment for respirable particles.

We have used the concentration 5.44×10^{-3} Bq m^{-3} as the geometric mean of a lognormal distribution with geometric standard deviation 16 (two orders of magnitude between the 50th and 95th percentiles). This is consistent with the range of most of the tabulation given in Table 12-7 of Sehmel (1984). The fifth percentile is about twice the median value for grass cover, which we have taken as the median of a lognormal distribution with geometric standard deviation 4, derived previously. Thus, there is some overlap of the two 90% probability intervals (Figure 5-6).

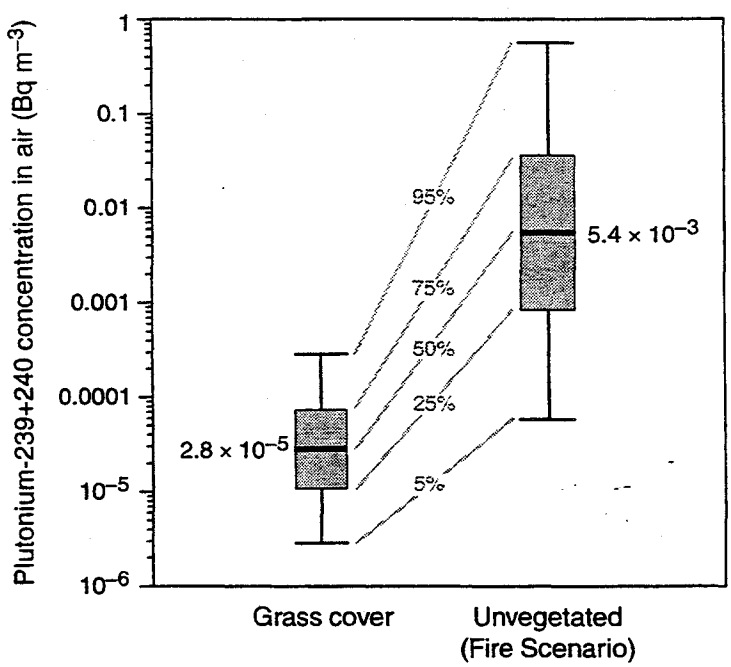

Figure 5-6. Predicted concentration of $^{239+240}Pu$ in air at Rocky Flats for uncut grass and unvegetated soil in the wake of a hypothetical fire. The location is 300 m east of the center of the 903 pad. The percentiles of the uncertainty distributions are indicated in shaded type. The

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distributions are lognormal, with geometric means (50th percentiles) as shown and geometric standard deviations 4 and 16 for grass cover and unvegetated ground, respectively.

5.4 Resuspension Parameters for the Scenarios

Table 5-4 gives air concentrations for $^{239+240}Pu$ at the principal scenario locations. These air concentrations correspond to the flux estimates derived in Sections 5.2 and 5.3, together with the distribution of $^{239+240}Pu$ in soil that is based on interpolation in site data (Section 4.1). Table 5-4 shows the soil concentration at the principal location of each scenario, but we remind the reader that the estimate of air concentration due to resuspension depends on a wide spatial distribution in the soil, and not only on the point value given in the table. Receptor points are moved to place them where air concentrations or air-to-soil concentration ratios are maximum (e.g., Section 7).

Table 5-4. Plutonium-239+240 Air and Soil Concentrations for Rocky Flats Scenarios

Scenario	Distance (km)	Angle (°)	Air ($Bq\ m^{-3}$) Uncut grass	Air ($Bq\ m^{-3}$) Unvegetated soil	Soil ($Bq\ kg^{-1}$)
Site workers	0.59	330	5.6×10^{-7}	9.9×10^{-5}	17
Rancher, DOE resident, DOE recreational user	0.30	90	2.8×10^{-5}	5.4×10^{-3}	8.9×10^{3}

5.5 Probabilistic Incorporation of Future Fires into the Scenarios

A wildfire on the Rocky Flats site is not necessarily a natural occurrence (fires may be unintentionally or deliberately set by people), but it is an event that is beyond ordinary control. We do not consider wildfire occurring on the site as part of a scenario definition. Rather, it is part of the exogenous environment that should be treated probabilistically, just as we treat processes such as resuspension and radionuclide concentrations in air. With the 1000-year temporal scope of the assessment, the probability of occurrence of one or more wildfires that would call for the flux parameters derived in Sections 5.3 and 5.4 is not negligible. In this section, we outline a method of estimating RSALs for a scenario, with probabilistic consideration of future wildfire at the site.

5.5.1 Fundamental formulas

Fire statistics for this century in the Arapaho and Roosevelt National Forests (1.3 million acres, 5.3×10^9 m^2) and the Pawnee National Grasslands (193,000 acres, 7.81×10^8 m^2) are available from the Fort Collins Interagency Wildfire Dispatch Center (web address http://www.fs.fed.us/outernet/arnf/welcome.html). Fire events within these Northern Front Range regions that have burned 100 acres (4.05×10^5 m^2) or more are summarized in Table 5-5 (we will use the term "large fire" to mean a fire of at least this magnitude). We use these statistics as a basis for simulating wildfire events at the Rocky Flats site within the 1000-year temporal scope of the assessment.

We choose a discrete probabilistic model of fire occurrence, with each of the 1000 years eligible for either the occurrence or non-occurrence of a fire (a *fire event*). Fire events in different years are assumed stochastically independent. Multiple fires in the same year are not considered. We assign a parameter p as the probability that one fire will occur in any specified year, and we assume that the value of p is the same for all years. (The latter assumption, of course, does not take into account long-term fluctuations of precipitation and drought that might be expected in a millennium, nor changes in human incursion that would affect rates of accidental or deliberate fire setting.)

Using the data in Table 5-5, we can estimate a value for p. Application of this value to the 1000-year period is obviously an extrapolation, since the data are confined to the Twentieth Century.

The area of the ranch assumed for the *RAC* Rancher scenario is 1×10^7 m^2, and we take this rectangular region as the domain of the fire. Its extent includes the contours of highest plutonium concentration in the soil, and it is the location of primary concern for such a fire. The ranch is bounded on the east by Indiana Street, on the north partly by an inner security fence just south of Highway 128, on the south by the site boundary, and on the west by a north-south line just west of the 903 pad. The southwest and northeast corner coordinates that define the fire domain are taken as (483.20, 4413.12) and (485.96, 4417.23) km UTM, respectively.

Table 5-5 indicates that from 1900 through 1998, there were 54 large fires in the Arapaho and Roosevelt National Forests and the Pawnee National Grasslands (total area 6.1×10^9 m^2). However, the large gap from 1900 to 1924, in contrast to shorter gaps after 1932, suggests that the earlier record may be lacking unreported fires. It is also possible that the frequency of fires has increased as more people have gained access to these parklands. A large fraction of the fires are attributed to human causes, and we consider the period from 1924 through 1999 more appropriate as a temporal scope for computations with the fire statistics. Taking the Arapaho and Roosevelt National Forests and the Pawnee National Grasslands as our calibration domain, we estimate the annual fire frequency as

$$54 / 75 = 0.72 , \tag{5-15}$$

which can be interpreted as an estimate of the probability that a large fire will occur somewhere in these parklands in any specified year, assuming that these statistics would be representative of conditions affecting fires during the year in question.

Given the occurrence of a large fire in the parkland domain, the probability that it will be located in a specified subregion R is the area of R divided by the total area of the domain (6.1×10^9 m^2). The product of this probability and the frequency 0.72 (Equation 5-15) is the probability of the occurrence of a fire in R in any specified year. If we assume that the same areal rate can be applied to the land of the *RAC* Rancher, then we may estimate the probability p as

$$p = \frac{1 \times 10^7 \text{ m}^2}{6.1 \times 10^9 \text{ m}^2} \times 0.72 = 1.2 \times 10^{-3} . \tag{5-16}$$

This is a little more than one fire per thousand years on the rancher's land. We note that there were 7 large fires in the Pawnee National Grasslands, and all were recorded after 1960. If that park's area and large-fire frequency are substituted in Equation (5-16), the probability estimate is 2.3×10^{-3}, about twice the value in Equation (5-16). The Pawnee National Grasslands might be considered a better surrogate for the Rocky Flats site than would the Arapaho and Roosevelt National Forests, but the small number of observations gives a poorer representation of the probability distribution of burn areas than the combined grasslands and forests. We adopted the

value 1.2×10^{-3} (Equation 5-16) as a minimum value of the parameter p, with maximum 2.3×10^{-3}. Sampling was from a log-uniform distribution, with median 1.7×10^{-3}.

Table 5-5. Recorded Wildfires >4.05×10^5 m^2 (100 Acres) in the Arapaho and Roosevelt National Forests and Pawnee National Grasslands 1900–1998[a]

Year	Area (km^2)	Cause	Year	Area (km^2)	Cause
1924	4.65	Unknown	1971	0.85	Human
1932	1.82	Human	1976	1.04	Human
1932	2.02	Railroad	1976	1.01	Human
1934	1.58	Railroad	1978	1.62	Human
1938	2.02	Railroad	1978	4.86	Human
1939	4.08	Human	1979	1.26	Human
1943	0.93	Human	1980	1.93	Human
1944	4.82	Lightning	1980	11.06	Human
1944	0.69	Lightning	1980	0.73	Human
1952	0.72	Human	1980	4.05	Lightning
1952	4.91	Lightning	1985	0.95	Lightning
1953	1.15	Human	1985	1.21	Lightning
1954	1.04	Human	1988	11.33	Human
1956	2.89	Lightning	1988	3.06	Human
1958	0.68	Lightning	1988	10.00	Human
1960	2.45	Human	1988	0.81	Lightning
1960	2.67	Lightning	1989	7.30	Human
1960	0.61	Lightning	1989	1.47	Lightning
1962	4.08	Human	1989	7.96	Lightning
1962	3.03	Human	1989	0.40	Human
1962	0.57	Human	1990	0.57	Lightning
1963	0.74	Human	1993	1.11	Lightning
1963	0.89	Lightning	1994	4.94	Lightning
1966	1.90	Human	1994	1.50	Lightning
1968	2.99	Lightning	1996	1.50	Lightning
1971	10.66	Human	1998	0.53	Human
1971	0.50	Lightning	1998	1.93	Human

[a] Source: Fort Collins Interagency Wildfire Dispatch Center, http://www.fs.fed.us/outernet/arnf/fire/fire.html

In any of the RAC Rancher scenarios, the probability that no fire will occur on the Rancher's land during the 1000-year temporal scope is $(1-p)^{1000} = 0.18$, and the probability that at least one fire will occur there in that period is 1 – 0.19 = 0.82. A probability of this magnitude is not negligible, but even if a fire occurs, it is not necessarily true that the fire would produce the maximum annual dose to the contemporary scenario subject who returns to the land immediately afterward, or that the dose from exposure to the burned area would exceed the 15 mrem annual limit. It is possible that a subject exposed to the unburned site in the year 2000 would receive a

larger annual dose. Which of the exposed subjects would receive the maximum annual dose depends on the date, nature, and location of the fire. In the next subsection, we formulate a simulation method that considers the probability of fire and its effect on the subject's exposure to radionuclides in the soil.

5.5.2 Simulation of plutonium exposure that includes the possibility of fire

The worst-case conditions for a wildfire affecting the RAC Rancher scenarios (assuming that the subject escapes from the site at the time of the fire and does not breathe contaminated smoke) would include the following: (1) The fire would occur in the earliest year considered (nominally 2000). (2) The fire would devegetate the entire ranch land of the RAC Rancher scenarios. (3) Regrowth of the vegetative cover would require a year or more. If the scenario subject is assumed to return to the ranch immediately after the fire, these conditions would give the maximum exposure, which would occur during the year of the subject's tenancy on the bare soil. But the probability that all three conditions will coexist is small, and a more realistic treatment requires explicit consideration of their possible mitigation.

The smallest fire considered (burn area of 4.05×10^5 m^2 or 100 acres) would devegetate only about 4% of the rancher's land. Such a bare subplot could be located 1–2 km from the parts of the ranch where the residents spend most of their time, possibly in a direction that is not often upwind from the residents, and possibly where plutonium concentrations in soil are relatively low. In addition, the fire could occur several centuries into the future, leaving time for weathering processes to have removed a significant fraction of the plutonium from the surface layer of the ranch soil, thus making less plutonium available for resuspension. Such mitigating factors need to be considered, at least approximately, in simulations that consider the possible role of a fire in exposing the subjects to plutonium from the soil. Even if a fire is assumed to occur sometime in the future (or even in the first year considered), if mitigation is taken into account, the fire may not produce an annual dose exceeding the annual dose that a subject exposed to the unburned site during 2000 would receive.

The purpose of the simulation scheme outlined below is to extend the existing scenarios to include a probabilistic consideration of a fire on the land associated with the RAC Rancher scenario, where most of the plutonium in soil is located. First, we describe the parameters and random variables in the simulation.

The fundamental parameter is the probability p that a fire will occur in any specified year. In Section 5.5.1, we adopted a distribution with median value $p = 1.7 \times 10^{-3}$.

The objective of the simulation is the ratio Φ of the resuspension flux r_F of soil from a burned region and the resuspension flux r_G with the contemporary (year-2000) grass cover:

$$\Phi = r_F / r_G . \tag{5-17}$$

The baseline value of Φ is Φ_0, the approximate worst-case ratio given in Sections 5.3 and 5.4.

The first modifying factor is the reduction of the fire resuspension flux to account for weathering of the soil. This factor is $\exp(-\alpha(m-1))$, where α (year^{-1}) is the weathering rate and m is the year in which the fire occurs ($m = 1$ corresponds to 2000). We describe the Monte Carlo estimation of m below.

The second modifying factor, f_B, is related to the randomly determined size and location of the fire. It is proportional to the air concentration of plutonium at the scenario subject's principal location resulting from resuspension of soil within the burn region. The proportionality constant

is chosen so that the highest concentration (i.e., the worst case) corresponds to $f_B = 1$. For a simulated fire, the burn region is assumed circular, with area determined by sampling of burn areas from Table 5-5 (all areas have equal probability of being chosen) and dividing that area by the area of the fire domain (equal to the area of the ranch). The center of the circular region is constrained to lie inside the fire domain, but part of the burn region may lie outside. The resuspension flux is proportional to the average plutonium concentration in the soil within the burn region. The concentration of airborne plutonium at the receptor location is estimated with a Gaussian plume model for a circular area source, considering average wind speeds for stability class D and 16 wind-directional frequencies.

The final modifying factor g represents mitigation due to the recovery time of the vegetation. We assume that this recovery time is between 6 months and one year and represent it as a uniformly distributed random variable determined by the extreme values 0.5 and 1. In periods of extreme drought, longer recovery periods might be possible, but for the maximum annual dose criterion prescribed for this project, they would not be relevant.

Here is how the simulation proceeds:

(1) Set flux ratio $\Phi \leftarrow 1$ and the fire year $m \leftarrow m^* \leftarrow 1$ (m^* will keep track of the year in which the maximum flux ratio occurs).
(2) Sample a number u from the uniform distribution on [0, 1]. If $u \le p$, we have a fire in year m, so go on to step (3). Otherwise, go to step (4).
(3) Generate f_B and g independently as described above. Set $\varphi \leftarrow \Phi_0 f_B g \exp(-\alpha(m-1))$. If $\varphi > \Phi$, set $\Phi \leftarrow \varphi$ and $m^* \leftarrow m$.
(4) If $m < 1000$, $m \leftarrow m+1$ and return to step (2).
(5) Otherwise, make the adjustment $\Phi \leftarrow \Phi \exp(\alpha(m^*-1))$ and exit. [This adjustment is carried out because RESRAD will calculate the weathering loss of plutonium from the surface soil. If we left the exponential factor in place, the effect of weathering would be counted twice in the RSALs.]

The effect of this sequence is to find the maximum value of Φ over all fires that occur within the 1000-year period. If at the end the flux ratio $\Phi = 1$, we conclude that either no fire occurs or else the fires that do occur have resuspension fluxes that do not exceed the year-2000 flux with normal vegetation.

The above sequence is embedded in a larger Monte Carlo process that invokes RESRAD with different sets of parameters and collects the results as empirical distributions. When Φ is generated ($1 \le \Phi \le \Phi_0$), we use it to multiply the geometric mean of the plutonium air concentration corresponding to the year-2000 grass cover on the site. If $\Phi = 1$ (no "significant" fire), there is no change in the geometric mean, and the uncertainty factor (geometric standard deviation) for ordinary grass cover is used (Sections 5.1 through 5.2). If $\Phi > 1$ (a "significant" fire), multiplication by Φ increases the geometric mean of the air concentration to correspond to resuspension from unvegetated soil, and the geometric standard deviation for the fire resuspension flux is used (Sections 5.3 and 5.4).

Application of the fire to scenarios other than the *RAC* Rancher is straightforward, except that the same fire domain is used, even though the scenario subject's primary location may be outside of the fire domain.

6. URANIUM METHODOLOGY

RSALs were calculated differently for uranium and plutonium because the nature and extent of contamination differed between the nuclides. Our treatment of plutonium considered a 10-km^2 contaminated area. Using spatially-variable soil concentrations and measured air concentrations of plutonium around the site, we calibrated a suspension model so that the suspension rates of plutonium-contaminated soil would yield concentrations currently measured at the air samplers.

This procedure was not extended to uranium because (a) uranium-specific measurements were not available at the samplers, and (b) uranium contamination is not as widespread as plutonium and, therefore, would not be expected to respond in the same manner. Our investigation indicated that uranium contamination was mainly limited to past disposal areas and burn pits. Furthermore, Litaor (1995) noted fundamental differences in solubility characteristics of plutonium and uranium that, in turn, affected their mode of dispersion in the environment. Litaor (1995) reported uranium isotopes to have migrated down to about 40 cm below the soil surface, indicating greater aqueous-phase mobility for uranium compared to plutonium. Migration of uranium below the soil surface reduced its susceptibility to dispersion by wind suspension. Therefore, uranium contamination was restricted to areas where it was originally introduced to the soil, and it tended to only move vertically down the soil column.

Because uranium contamination was limited to discrete areas on the RFETS, we treated the source as a hot spot and restricted its area to 100 m^2. This area differs from what was assumed in the DOE/CDPHE/EPA methodology, which assumed a 40,000-m^2 area.

6.1 Uranium Mass Loading

To address the resuspension pathway from uranium-contaminated areas, we applied the mass loading factor approach used in RESRAD. As stated in the previous paragraph, we could not use the calibrated air model used for plutonium for uranium. The mass loading factor relies on dust loading measurements and was the approach DOE used to calculate RSALs (DOE/EPA/CDPHE 1996). The air concentration from resuspension was estimated using

$$C_a = ML\,C_s \tag{6-1}$$

where

$C_a =$ airborne concentration from resuspension

$ML =$ mass loading factor (g m^{-3})

$C_s =$ surface soil concentration (Ci g^{-1}).

The air concentration was further modified by an area factor that accounted for dilution of the airborne mass from uncontaminated dust that entered the airstream upwind from the source. The area factor approaches 1.0 for very large areas. The original RSALs (DOE/EPA/CDPHE 1996) were based on a mass loading factor of 26 μg m^{-3}. Mass loading in the vicinity of Rocky Flats has been measured by the CDPHE, and Hodgin (1998) provided a review of these data for 1995, 1996, and 1997. The annual average geometric mean (GM) total suspended particulates (TSPs) was 37 μg m^{-3} at the east end of the Industrial Area and 27 μg m^{-3} in the interior of the east Buffer Zone. Geometric mean TSP concentrations around the perimeter of the RFP ranged

from a high of 39.8 μg m^{-3} at the west perimeter to a low of 25.6 μg m^{-3} at the northern perimeter of the site.

Mass loading factors were reviewed for Phase II of the Rocky Flats Dose Reconstruction project in Rood and Grogan (1999). Raw data from the perimeter monitoring stations were obtained from personnel at CDPHE for the years 1992–1998. The GM of the six annual average TSP concentrations was 35 μg m^{-3}, with a GSD of 1.25. In the Phase I Dose Reconstruction Project, ChemRisk reported the GM TSP annual average concentration at Rocky Flats was 60 μg m^{-3} for 1980, 1983, and 1984 (ChemRisk 1994). This value was obtained verbally from CDPHE personnel,[a] and records of this information were not obtained. We queried CDPHE about this value, and they referred us to the later measurements that were considerably lower than those used by ChemRisk in Phase I. A mean of 35 μg m^{-3} appears more reasonable than 60 μg m^{-3} based on other measurements near Rocky Flats and elsewhere. According to the U.S. Environmental Protection Agency (EPA), the annual mean TSP concentration at 30 nonurban sites ranges from 5 to 50 μg m^{-3} (EPA 1990). Whicker and Schultz (1982) gives an average dust loading in nonurban locations of about 40 μg m^{-3}. For urban locations, Gilbert et al. (1983) reports TSP concentrations range from 33 to 250 μg m^{-3}.

Because we are dealing with a nonurban environment, we believed it was more appropriate to use the site-specific mass loading factors. Therefore, we applied a mass loading factor having a GM of 35 μg m^{-3} and a GSD of 1.5. The GSD was increased to envelop the range of measured dust concentrations at the RFP.

Not all particulates are respirable. Particulate matter less than 10 μm aerodynamic equivalent diameter (PM-10) is generally considered in the respirable size range. Hodgin (1998) reported that PM-10 concentrations ranged from 30 to 40% of the TSP concentrations. The PM-10 concentration is then the TSP concentration multiplied by the fraction of the TSP that is <10 μm. Therefore, we assigned a uniform distribution to the fraction of TSP that is <10 μm having a minimum of 0.3 and a maximum of 0.4.

6.2 Implications of a Fire

Section 5.5 discusses the possibility of a fire sometime in the future at the RFETS. Using fire statistics for this century in the Arapaho and Roosevelt National Forests and the Pawnee National Grasslands, the probability of a fire occurring on the ranchers land at the RFETS was estimated to be 9×10^{-4}. The smallest fire area considered in the fire statistics data set was 4.05×10^5 m^2 or 100 acres. Using the area encompassed by uranium contamination (100 m^2) in Equation (5-16) yields a probability of

$$p = \frac{1\times10^2\ \mathrm{m}^2}{6.1\times10^9\ \mathrm{m}^2}\times0.55 = 9.0\times10^{-9}.$$

This probability is 5 orders of magnitude lower than that for the plutonium case. Additionally, only the inhalation pathway is affected by the fire, and inhalation doses made up a small fraction of the total dose (Figure 6-1). Nevertheless, we ran a trial fire case to verify that even if there were a fire, the doses would not be significantly higher. For this trial, we conservatively assumed that any fire occurring on the site encompassed a uranium-contaminated

[a] Personal communication with N.D. Chick, Colorado Department of Health, Air Pollution Division, June 19, 1992.

area. In addition, the modifying factors described in Section 5.5.2 were not considered here. The mass loading factor was scaled according to the increase in the mass loading as observed by Anspaugh et al. (1975) and discussed earlier in this report. Anspaugh predicts a resuspension flux (6.0×10^{-3} mg m^{-2} s^{-1}) for sparsely vegetated soil that exceeds our estimate for Rocky Flats with its uncut grass (2.76×10^{-5} mg m^{-2} s^{-1}) by a factor of more than 200. Therefore, during the occurrence of a fire, we increased the mass loading by a factor of 200.

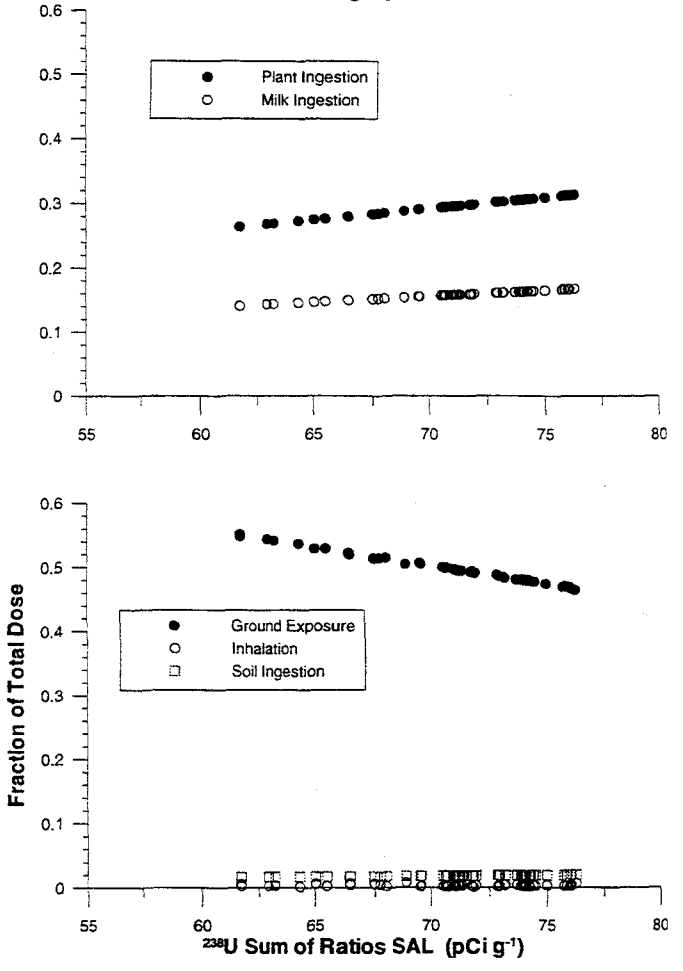

Figure 6-1. Fraction of the total dose at the time of maximum dose for different pathways of exposure as a function of the ^{238}U RSAL with the groundwater pathway turned off. The sum-of-ratios calculation is incorporated into the ^{238}U RSAL. Note that soil ingestion and inhalation make up only a small fraction of the total dose.

The fire calculation for uranium used the same statistics used in the plutonium simulations. That is, we calculated the year when a fire of greater than 100 acres would occur somewhere on the rancher's land. We essentially assumed that that fire would encompass an area contaminated by uranium. We performed the calculation using the mass loading factor for vegetated surfaces. Next, we extracted the doses from RESRAD at the time of the fire and at the time of maximum dose estimated by RESRAD from the SUMMARY.REP file. Because the inhalation dose is proportional to the mass loading factor, we scaled the inhalation dose according to the increase in

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the mass loading factor and recalculated the total dose at the time of the fire. The dose at the time of the fire was then compared with the maximum dose calculated by RESRAD and the larger of the two was selected. We ran this procedure for the rancher's child scenario and found that in all cases, the maximum dose calculated by RESRAD was higher than the dose at the time of the fire.

Several factors impacted this outcome. First, inhalation doses only accounted for a few tenths of a percent of the total dose. Second, the fire occurred sometime in the future and not at the year 2000. This delay allowed much of the uranium to leach below the soil surface where it was unavailable for resuspension. Third, unlike plutonium, the groundwater pathway was significant for uranium (see Figure 6-2). We concluded that incorporating a fire into the uranium calculations made no difference in the overall results. As shown in Section 11, including the groundwater pathway in the scenario played a greater importance for the uranium calculations.

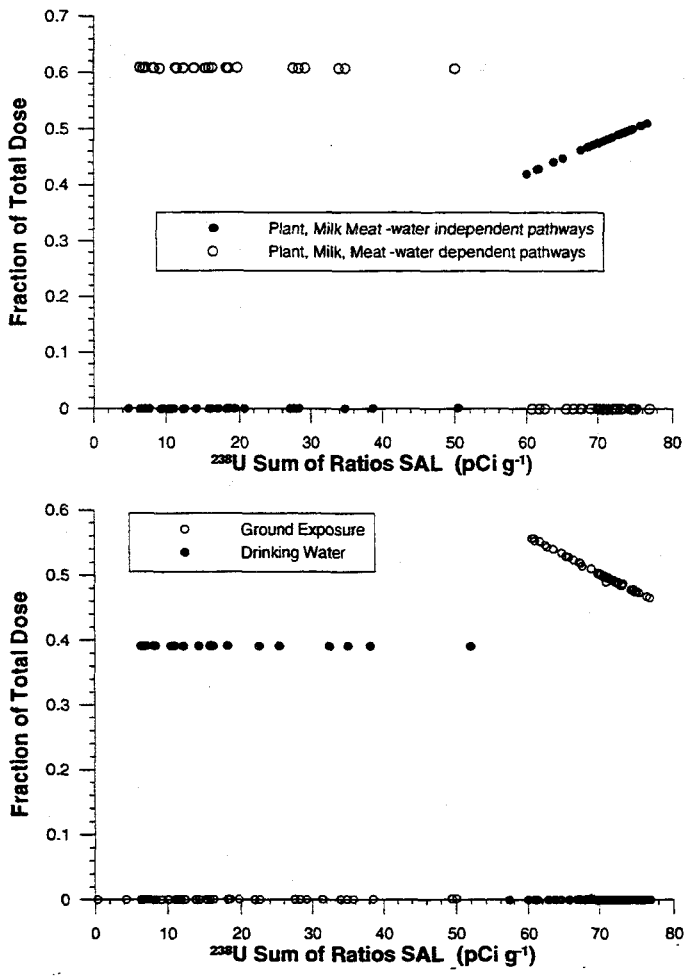

Figure 6-2. Fraction of the total dose at the time of maximum dose for different pathways of exposure as a function of the ^{238}U RSAL with the groundwater pathway turned on. The sum-of-ratios calculation is incorporated into the ^{238}U RSAL. Note that ground exposure is important at RSAL values >60 pCi g^{-1}. At RSAL values <60 pCi g^{-1}, groundwater-dependent pathways dominate. Groundwater becomes an insignificant pathway when the transit time through the unsaturated zone exceeds the 1000-year time of compliance.

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6.3 Sum-of-Ratios for Uranium

As was done with the plutonium isotopes, we also performed a sum-of-ratios calculation for the uranium isotopes. The sum-of-ratios calculation required that we know the relative abundance of each isotope in the soil. Isotopic ratios were based on data in Litaor (1995) in his examination of Disposal Pits 1–4, 8, 9, 11, and 12. We were concerned with three isotopes of uranium; ^{238}U, ^{235}U, and ^{234}U. Uranium-238 and ^{234}U were assumed to have the same activity—that is, they had an isotopic ratio of 1. Isotopic ratios for ^{235}U ($^{238}U/^{235}U$) were assigned a log triangular distribution having a minimum of 6.2, a most likely value of 25, and a maximum value of 92.

We kept the sum-of-ratios calculation for uranium separate from that of plutonium. It was not possible to combine to two for a generic site because uranium contamination is localized and not widespread as plutonium is. We would have to apply a sum-of-ratios calculation case-by-case after the activity levels for each isotope were determined.

7. AN ALTERNATE RSAL CALCULATION METHOD

7.1 Truncation of Soil Concentration at a Specified Contour

In Section 5.2.1, we mentioned the contour truncation feature of the air transport and diffusion model, in which all soil concentrations within a specified contour can be given the value of the contour (i.e., we truncate the concentrations to that value). For example, if we wish to simulate remediation to a maximum of 4000 Bq kg^{-1} (108 pCi g^{-1}), the model levels all concentrations within the contour line defined by this value to 4000 Bq kg^{-1} (108 pCi g^{-1}). Thus, the simulation treats this value as the maximum concentration for the site. This capability makes it possible to estimate RSALs that correspond more explicitly to the specified conditions than RSALs calculated with the sum-of-ratios scheme used elsewhere in this report. We refer to this approach as the alternate RSAL calculation method.

Using the contour truncation feature of the model, we can estimate RSALs directly by calculating maximum annual dose for various maximum contour values, as if remediation had been carried out as indicated by those values. When we reach a contour concentration that corresponds to the prescribed dose limit, that concentration is the RSAL. Since we are applying uncertainty techniques, each contour value generates a dose distribution. In this context, we choose as the RSAL the contour value for which the limiting annual dose is approximately the 90th percentile of the dose distribution, in accordance the principle that the probability of exceeding the dose limit should be approximately 10%.

This alternate method, which we describe below by means of an example, is somewhat more difficult to implement than the approximation methods we have applied to generate the probability curves in Section 11. It is not easily automated, and thus it is not yet suitable for generating the full range of RSALs for all scenarios and variants. However, because it is more explicit, it is useful as a check on the sum-of-ratios method. A careful scrutiny of it may help to clarify some questions connected with the definition of a soil action level in heterogeneous media such as those that exist at the Rocky Flats site.

7.2 Modification of the Fire Model

This calculation requires a modification in the fire model to enable it to account for the truncated concentration within the maximum contour for those cases when the burn region intersects the contour region. It was necessary to make use of absolute rather than relative soil concentrations, and the estimated fire/no-fire flux ratio is of a different form. In the remainder of this section, we give some details of the modified fire model.

The modification applies to the material of Section 5.5.2. We still estimate the flux ratio Φ of Equation (5-17), and we calculate the air concentration due to resuspension from the randomly generated circular burn region with a circular area-source Gaussian plume model, but now we take into account the truncation within the burn region if that region intersects the maximum contour. The total flux from the circular region is $r_0 = r_{2000} + \Delta r$, where r_{2000} is the baseline soil flux with normal vegetation, and Δr is the incremental flux due to the devegetation. We denote the unmodified air concentration by C_a, the concentration component per unit flux from the burn region by χ (calculated by the circular area-source Gaussian plume model), and the modified air concentration at the time of the fire by C_a^*. Then we may write

$$C_a^* = g\,(C_a + \chi \cdot \Delta r)\exp(-\alpha t) = C_a g\,(1 + \chi \cdot \Delta r / C_a)\exp(-\alpha t) = C_a \Phi \qquad (7\text{-}1)$$

where, as in Section 5.5, g denotes mitigation due to vegetation regrowth, and α is the first-order weathering coefficient for leaching of plutonium; t is the number of years after 2000 that the fire occurs. The symbol Φ represents our estimate of the plutonium activity flux ratio (the corresponding quantity without the exponential factor is the soil mass flux ratio). If $\Phi > 1$, then $C_a^* > C_a$, and the fire is considered significant, which means that we use the dose estimate based on the future fire rather than the dose estimate based on the vegetated site in the year 2000. For each Monte Carlo realization, this choice must be made.

Here is an algorithmic outline of the calculation of the plutonium flux ratio:

(1) Set flux ratio $\Phi \leftarrow 1$ and the fire year $m \leftarrow m^* \leftarrow 1$ (m^* will keep track of the year in which the maximum flux ratio occurs). C_a and Δr are already available.
(2) Sample a number u from the uniform distribution on [0, 1]. If $u \le p$, we have a fire in year m, so go on to step (3). Otherwise, go to step (4).
(3) Generate χ and g independently. Set $\varphi \leftarrow (1 + \chi \cdot \Delta r / C_a)\exp(-\alpha(m-1)) \cdot g$. If $\varphi > \Phi$, set $\Phi \leftarrow \varphi$ and $m^* \leftarrow m$.
(4) If $m < 1000$, $m \leftarrow m + 1$ and return to step (2).
(5) Else set $t \leftarrow m - 1$ and exit.

We use the criterion $\Phi > 1$ to indicate the occurrence of a significant fire. If one occurs, t indicates the year. The modified mass loading factor in RESRAD must be based on the soil flux ratio rather than the plutonium flux ratio. Accordingly, when Φ is used for this purpose, it must be multiplied by the factor $\exp(\alpha t)$.

7.3 Calculations with the Alternative Method

We have applied the contour truncation feature of the air transport and diffusion model to the *RAC* Rancher scenario, where the subject resides on-site. We begin with a high maximum contour value and work downward until the 15 mrem year^{-1} dose limit occurs near the 90th percentile of the maximum annual dose distribution. Table 7-1 shows the results as the maximum contour value is reduced. In each case, the subject is assumed to be situated within the maximum contour, on the eastward radial from the 903 pad, at the point of maximum air concentration (this point is determined experimentally by calculating air concentrations at intervals of 0.1 km). Note that the subject location moves eastward as the maximum contour value decreases (and hence the remediation criterion grows more stringent). Note how the percentile at which 15 mrem year^{-1} occurs increases. The values in Table 7-1 are based on 200 Monte Carlo realizations per simulation. The uncertainty of the soil concentration is represented by a lognormal distribution with geometric standard deviation (GSD) 2, corresponding to a 90th/50th percentile ratio of 3, which is approximately the generic guidance for applying RESRAD. We mention this choice of the GSD because the RSAL estimate is sensitive to this assumption.

The last two lines of Table 7-1 place the $^{239+240}$Pu RSAL between 27 and 54 pCi g^{-1}. Linear interpolation gives the value 38 pCi g^{-1}. This result includes simulation of future fires within the fire domain as indicated in Section 7.2. To provide a sense of the importance of the fire simulation to the alternate method with this scenario, we repeated the calculation with the

probability of a fire in any specified year set equal to zero; this setting prevents all fires. The RSAL for this no-fire simulation is 44 pCi g^{-1}. Thus the fire, as implemented, has a small effect on this simulation. However, there are technicalities in the method of implementing the fire model that might change its relative importance under different algorithmic arrangements. These technicalities are related to making uncertainty estimates for the plutonium soil concentrations compatible between the master script and the external program that implements the fire model. We have had no opportunity to explore this question and cannot elaborate on it here.

Sensitivity to the fire for the extended sum-of-ratios method depends on the air-to-soil concentration ratio at the receptor location. For the rancher scenario, it ranges from about a factor of two (with the higher RSAL corresponding to no fire) at the eastern site boundary to a negligible effect near the 903 pad. Receptors placed at the eastern margins of the site are more sensitive to the occurrence of a fire because they tend to be downwind from the fire locations.

The alternate calculation's value of 38 pCi g^{-1} might be expected to differ from the value given by the probability curve based on the extended sum-of-ratios method. The latter method is a linear extrapolation, which depends on $^{239+240}$Pu air-to-soil concentration ratios that are calculated from the unremediated spatial soil distribution of $^{239+240}$Pu on the Rocky Flats site. RESRAD-calculated RSALs are functions of location only through air-to-soil concentration *ratios* of $^{239+240}$Pu, but the RSALs are independent of soil concentration. Also, the receptor remains at a fixed location at the eastern site boundary, whereas the alternate calculation puts the receptor where the post-remediation air concentration is maximum. However, the similarity of results between the alternate calculation and the extended sum-of-ratios method (38 vs. 35 pCi g^{-1}) gives a degree of confirmation of the latter estimate. The near equality of these estimates should not be overemphasized: one could argue that the factor-of-three uncertainty assumed for the soil concentrations in the alternate calculation should be applied to the result. If this were done, we would have a 90% uncertainty interval of roughly 10 to 100 pCi g^{-1} about the alternate estimate. Even so, this interval does not differ drastically from the range discussed in the conclusions

Table 7-1. Accurate Estimation of $^{239+240}$Pu Soil Action Level for the RAC Rancher Living on the Rocky Flats Site, Using the Truncation Feature of the Transport and Diffusion Model

Maximum contour[a] concentration (Bq kg^{-1})	(pCi g^{-1})	Distance east from 903 pad[b] (km)	Air concentration (Bq m^{-3})	(pCi m^{-3})	Probability[c] that maximum dose exceeds 15 mrem y^{-1}
8000	216	0.3	1.28×10^{-5}	3.46×10^{-4}	85%
6000	162	0.3	1.05×10^{-5}	2.83×10^{-4}	70.5%
4000	108	0.4	7.72×10^{-6}	2.08×10^{-4}	51.5%
2000	54	0.5	4.58×10^{-6}	1.24×10^{-4}	20.1%
1000	27	0.6	2.62×10^{-6}	7.07×10^{-5}	3.4%

[a] Concentration of plutonium in soil inside the maximum contour is assumed to be uniform and equal to the contour value. This concentration is assumed to be known within a factor of 3, and this uncertainty is represented by a lognormal distribution with geometric standard deviation 2.
[b] Distance corresponds to maximum air concentration.
[c] These probabilities are based on interpolated percentiles of empirical distributions from 50 Monte Carlo realizations each.

A calculation with the alternate method for a location just east of Indiana Street, using the same rancher scenario and the current spatial distribution of $^{239+240}Pu$ in soil, indicated a dose distribution with the 15 mrem year^{-1} dose limit at approximately the 99th percentile. This result indicates that no remediation would be required to ensure with 99% probability that individuals at this location would not exceed the 15 mrem year^{-1} dose limit, provided they did not spend time on the site. Such a result would be applicable only in the case of a long-term institutional sequestration of the site.

7.4 Implications for the Definition of a Soil Action Level

This approach to calculating RSALs raises the question of exactly what is meant by such a hazard index. A general unambiguous definition is difficult to formulate for a conceptual model with heterogeneous concentrations in environmental media such as one required by the Rocky Flats site. The alternative calculation method just illustrated points to a definition related to a remediation strategy of creating a uniform truncated concentration within a specified maximum concentration contour line. Given this strategy, one can define the soil action level as the maximum-contour concentration that produces 15 mrem year^{-1} dose limit for the scenario being considered (or rather that produces a dose distribution having this dose limit as its 90th percentile).

But one can imagine situations in which the dose limit might be met by local remediation that would not reduce the global maximum concentration. In such a case, one would have to modify the definition to suit the site and the remediation strategy. For example, on a site with two high-magnitude maximum locations separated by some distance, it might be possible to meet the dose limit criterion for some exposure scenarios by reducing contamination near the lesser of the hot spots and leaving the other intact. Such a strategy would succeed without reducing the global maximum and would require a different definition of a soil action level. This example illustrates that a definition of the concept depends on the proposed dose reduction strategy. A weakness of the extended sum-of-ratios method we have used in this report is the difficulty of inferring a remediation strategy that corresponds to the result.

Because of ambiguities such as the one just illustrated, we have expressed reservations about the use of soil action levels and have suggested that less emphasis be placed on them. Calculation of the distribution of maximum annual dose (or any other criterion) for each of a sequence of remediation steps gives a constructive picture of the relationship between soil concentration levels and the goal to be met. Such calculations are well within the state of the art, and they need not be prohibitively expensive. We would recommend that they be tailored to the site under study, just as we have made these calculations specific to the Rocky Flats site.

8. THE ROLE OF RESRAD

Straightforward application of RESRAD to the Rocky Flats site encounters difficulties because of the highly nonuniform spatial distribution of $^{239+240}Pu$ in soil on the site. Using resuspension modeling that takes this large variation into account, we have estimated air-to-soil mass loading ratios (i.e., air concentration divided by soil concentration at the same location) that vary by more than an order of magnitude from one location to another. It is reasonable to apply RESRAD to the site if external models are first used to estimate a mass loading factor that is appropriate to the receptor location. It is also necessary to bypass the *area factor* that RESRAD calculates to account for dilution of contaminated airborne dust by uncontaminated dust from greater distances. This area factor is based on a simpler model of spatial distribution of contamination in the soil than is required for the Rocky Flats site.

We have used Monte Carlo calculations for the uncertainty analyses that we have carried out. Although a beta test version of RESRAD 3.81 with Monte Carlo capabilities was available when the study began, we preferred to control RESRAD with scripts written in PERL. The script invokes the computational module of RESRAD (RESMAIN3.EXE) repeatedly, using input files with randomly sampled parameter values, extracts the desired results from the output, and stores them for subsequent analysis. Some experimentation with the special version of RESRAD indicated that Monte Carlo simulations with it were likely to be significantly slower than with the scripting method. Moreover, the scripting approach offered some substantial conveniences for preprocessing data that the special version could not. Figure 8-1 summarizes the control cycle implemented by each PERL script. We prepared a separate PERL script for each scenario. A sample script is shown in Appendix A.

RESRAD embodies its own approach to resuspension. The approach is generically defensible for the models described by Yu et al. (1997), but it needs to be adapted for application to the Rocky Flats site. RESRAD estimates the air concentration of radioactivity C_a (Bq m^{-3}) as the product of a mass loading factor ML (kg m^{-3}), an area factor AF (dimensionless), and the concentration C_s of activity in surface soil (Bq kg^{-1}):

$$C_a = C_s \cdot \mathrm{ML} \cdot \mathrm{AF} \tag{8-1}$$

The mass loading factor may be estimated as the steady state mass concentration of resuspended soil particles. If the source area of the resuspended particles is effectively infinite and uniformly contaminated, then the air concentration is given by the product $C_s \cdot$ ML in Equation (8-1), so that AF = 1. Otherwise, the area factor is intended to adjust for the effect of dilution of the air concentration by uncontaminated soil particles that are transported from beyond the contaminated source region. It could be argued that other factors, such as variation of particle size distribution between the soil and the reference height, would also influence the area factor, but we accept the RESRAD interpretation for this discussion.

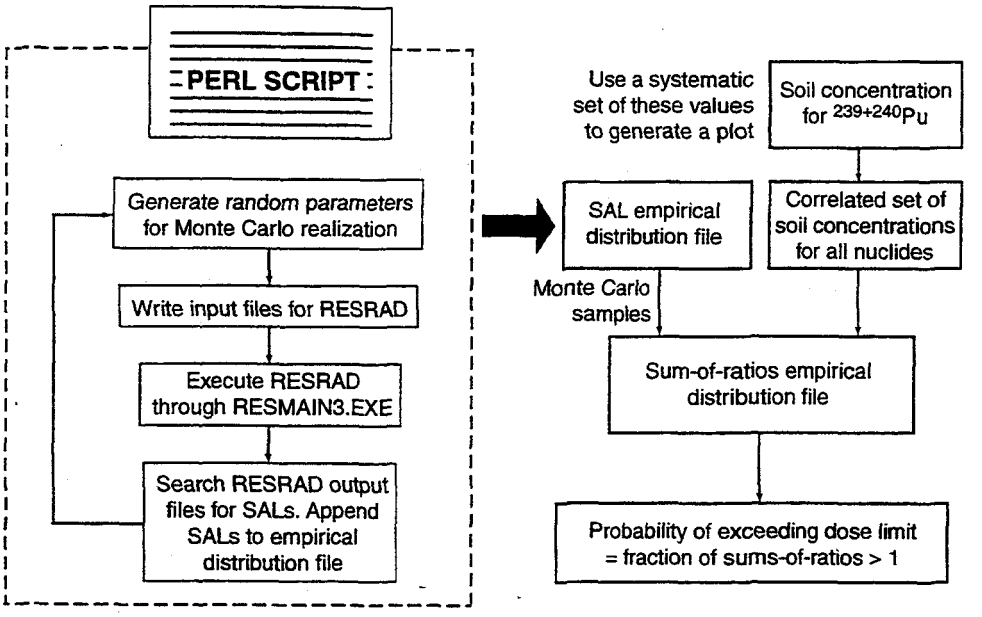

Figure 8-1. Schematic illustration of a scenario simulation using a PERL script with the RESRAD computational module. Prior calculations with other programs give a product ML · AF (mass loading factor times area factor) appropriate to the scenario and the spatial distribution of plutonium in the soil. Resulting Monte Carlo radionuclide soil action levels (RSALs) are collected in an empirical distribution file for postprocessing. Data for plotting are generated from the RSALs and a grid of soil concentration values for $^{239+240}$Pu. For each grid concentration, the calculated isotopic ratios are used to give a proportional set of concentrations of all the radionuclides. This set of radionuclide concentrations is combined with samples from the RSAL empirical distribution file (Equation 8-3), yielding an empirical distribution of sums-of-ratios. The fraction of these sums-of-ratios that are greater than 1 is the estimate of the probability of exceeding the dose limit, given the $^{239+240}$Pu soil concentration.

To understand the meaning of a RESRAD area factor for resuspension, one must consider a balanced process of suspension and deposition of uniformly contaminated soil that occurs upwind from a receptor location where we are interested in the air concentration. If the upwind fetch is infinite, we would anticipate a larger air concentration of radioactivity at the receptor point than would occur if the contaminated region were finite (which is what we are assuming in applications of RESRAD). The strategy in RESRAD is to estimate an air concentration that would correspond to an infinite region and adjust it by a factor that represents the ratio of concentration due to the finite area divided by the concentration due to an infinite fetch. Of course, a value equal to this ratio must be derived in a roundabout way, because the numerator of the ratio is the very concentration that one is trying to calculate. This ratio is called the *area factor* (AF) for resuspension.

The developers of RESRAD have derived an area factor that considers vertical and crosswind diffusion as represented by a Gaussian plume model, with gravitational settling estimated by Stokes's law (using a tilted plume to account for depletion) and wet deposition using a scavenging model. These models introduce additional parameters, such as the size distribution of aerodynamic diameters (1 to 30 μm is the size range considered in studying the variability of

the area factor), particle density, rainfall rate, raindrop size, wind speed, and the dispersion coefficients σ_y and σ_z as functions of atmospheric stability and distance from the source. The point source of the Gaussian plume is integrated over the finite contaminated area, while the receptor is kept fixed at the midpoint of the downwind boundary. The corresponding concentration for an infinite area is obtained by increasing the area of the square source region until the receptor concentration converges to a maximum value.

Reference values are assumed for some of the parameters, namely rainfall rate (100 cm year^{-1}), particle density (2.65 g cm^{-3}), atmospheric stability (Pasquill-Gifford class D, which typically occurs almost half of the time), and raindrop diameter (1 mm). The model is represented by a logistic regression curve, which was fitted to data generated by calculations for a grid of points in the parameter space. The function is

$$\mathrm{AF} = \frac{a}{1 + b(\sqrt{A})^c} \tag{8-2}$$

where A is the area of the contaminated zone and each of the parameters a, b, and c is a function of the particle diameter (μm) and wind speed (m s^{-1}). The functional correspondence for a, b, and c is shown in Table 4 of Chang et al. (1998).

Wind speed is available as an input to RESRAD, but particle aerodynamic diameter is not. The internal dose conversion factors for inhalation in the RESRAD database are based on activity median aerodynamic diameter 1 μm, and the RESRAD developers have chosen to fix the particle size parameter at this value for the present.

We replace the RESRAD product of the mass loading factor and area factor by a comparable quantity that is calculated outside RESRAD. This quantity is calculated as the ratio of the plutonium air concentration (Bq m^{-3}) and the plutonium soil concentration (Bq kg^{-1}) at the principal location of the scenario of interest. The air concentraton of radioactivity is calculated with the area-integrated Gaussian plume model, using the Rocky Flats soil contamination model. The ratio may be thought of as an estimate of the product ML · AF of an air-dust concentration and a generalized area factor that accounts for spatial variation of soil contamination in the source region. The product ML · AF represents an average over wind directions, wind speeds, and stability factors, whereas the interpretation of the RESRAD quantity considers only wind blowing across the contaminated fetch toward the receptor.

The RESRAD area factor $\mathrm{AF_R}$ is also calculated externally by the script program. By inputting the value ML · AF / $\mathrm{AF_R}$ for the mass loading parameter, we cause RESRAD to perform the calculation with mass loading factor ML and area factor equal to AF. The scripting program manages the calculation of the factor $\mathrm{AF_R}$ and substitution into the input file. Thus, the results are independent of the RESRAD area factor $\mathrm{AF_R}$. In this way, we adapt RESRAD to function as a module in a system that considers the extreme variation of soil concentrations and air-to-soil concentration ratios of plutonium on the Rocky Flats site (Figure 8-1).

We note that the approach of the RESRAD developers for the area factor of Version 5.82 might be interpreted (approximately) as a special case of the more general scheme we have followed for the Rocky Flats site. For applications conforming to the more restricted model structure addressed by RESRAD, we believe the approach described by Chang et al. (1998) is theoretically sound.

The Monte Carlo simulations shown in Figure 8-1 produce a file of RSALs for the plutonium, americium, and neptunium species of interest, one set of RSALs for each Monte Carlo realization (imagine a table with one row of RSALs for each realization and one column for each

radionuclide). In order to plot this information, we use a grid of values of $^{239+240}$Pu soil concentrations, one value for each point to be plotted. For each grid value (soil concentration), we sample the RSAL empirical file with replacement 1000 times (each sample would be one line from the imaginary table), and for each sample, we combine the individual nuclide RSALs that correspond to the $^{239+240}$Pu value in a sum of ratios:

$$S = \left(\frac{r_1}{\mathrm{RSAL}_1} + \cdots + \frac{r_N}{\mathrm{RSAL}_N} \right) C_{\text{Pu-239+240}} \tag{8-3}$$

where the subscripts point to the different individual isotopes ^{238}Pu, ^{239}Pu, etc., and $r_1, \ldots, r_N$ are the isotope activity ratios relative to the reference concentration of $^{239+240}$Pu. Thus, we would have $r_i C_{\text{Pu-239+240}}$ = concentration of nuclide i in the soil. The values $\mathrm{RSAL}_1, \ldots, \mathrm{RSAL}_N$ are the radionuclide soil action levels for a single Monte Carlo realization (i.e., the values from a single line of the imaginary table). The procedure generates a succession of random sums-of-ratios, $S^{(1)}, S^{(2)}, \ldots, S^{(B)}$. The probability that the dose limit will be exceeded can be estimated as the number of sums-of-ratios that exceed 1, divided by the total number B. (However, we have generally used interpolated values of cumulative probability rather than the discrete ratio.) The procedure is repeated for each grid value of the concentration $C_{\text{Pu-239+240}}$. The resulting probabilities are then plotted against the concentrations. These curves represent the probability of exceeding the dose limit as a function of $^{239+240}$Pu concentration in soil, and we usually refer to them simply as *probability curves*.

Note that for americium, neptunium, and the shorter-lived plutonium isotopes, the isotope activity ratios $r_1, \ldots, r_N$ should be specific to the time when the maximum dose would occur (Figure 3-1). However, we have always used the ratios for the year 2000, recognizing that this represents a distortion for those Monte Carlo realizations that correspond to a future fire event. The effect is likely to correspond to a higher dose estimate than would be the case for an event many years after 2000, when levels of the shorter-lived nuclides (^{238}Pu, ^{241}Pu, ^{241}Am) are lower relative to the longer-lived plutonium isotopes (^{239}Pu, ^{240}Pu, ^{242}Pu). However, ^{241}Am peaks in about 2032 (Figure 3-1), and until it returns to its 2000 level late in the 21st Century, the effect of using the isotope ratios for 2000 may slightly underestimate dose for fires occurring within that period. We do not consider this a matter for serious concern.

9. METHOD OF PROBABILITY PRESENTATION OF RSAL RESULTS

In this study, RSALs are presented in the form of a probability figure for each of the scenarios. Each scenario figure displays the probability of exceeding the dose limit as a function of the radionuclide concentration in the soil (in picocuries per gram). Figure 9-1 illustrates a generic probability curve. Each probability level corresponds to a distinct concentration of $^{239+240}Pu$ in soil. The probability value represents the probability of exceeding the dose limit; that is, at soil concentration A (picocuries per gram), there is a 5% chance that the person identified by the scenario will exceed the dose limit. Alternately, there is a 95% chance that the dose limit for the given soil concentration will NOT be exceeded. When we speak of probability levels throughout the report, we speak in terms of the probability of exceeding the dose limit.

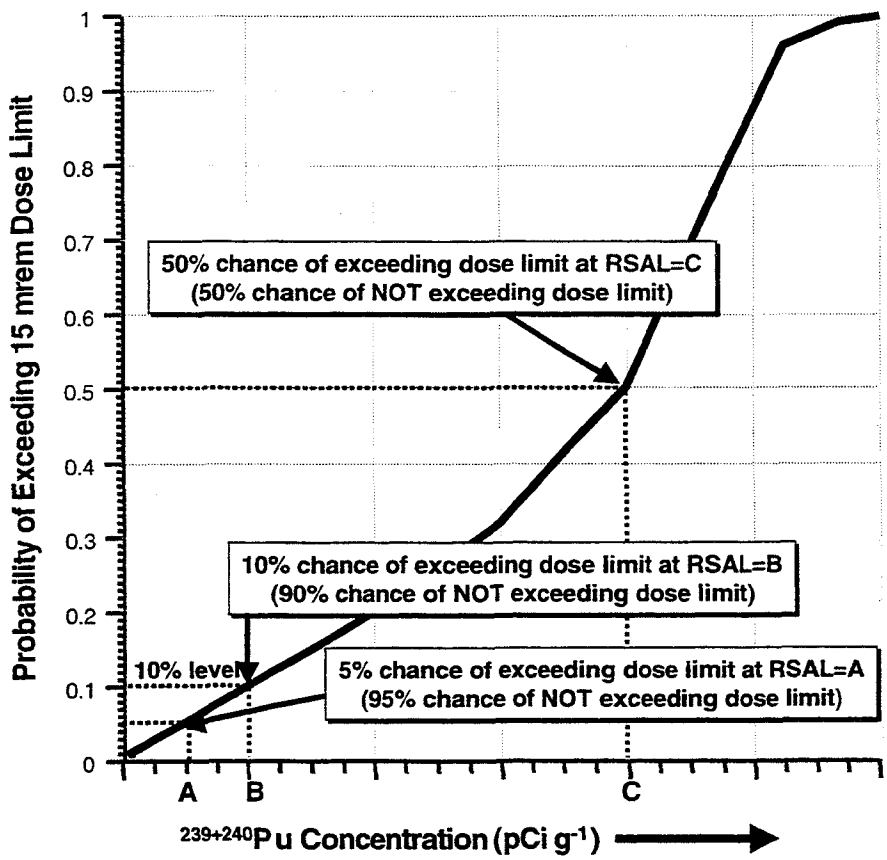

Figure 9-1. Generic distribution showing the relationship between probability of exceeding the dose limit as a function of plutonium concentration. As the contaminant concentration in soil increases, so does the probability of exceeding the dose limit.

10. CONSIDERATIONS IN SELECTING A RADIONUCLIDE SOIL ACTION LEVEL

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for the Rocky Flats site. In this report, we outline an approach to calculating RSAL values and provide an RSAL value that is supported by the scientific data, as specified in the scope of work. However, the final RSAL selection, which must consider additional aspects, ultimately lies in the hands of the stakeholders, DOE, and other State and federal authorities. Many criteria must be included in the ultimate selection of a soil action level, many of which cannot be taken into account in this analysis because they involve socio-political decisions outside the scope of our work. We list below some of the key criteria and explain those we have addressed and those that we have not addressed. Our approach should be viewed as a tool for selecting a scientifically defensible soil action level and a starting place for the value that is ultimately selected. Our work provides a firm technical basis on which a soil action level can be selected when these additional considerations have been taken into account.

Criteria that have been taken into account in *RAC*'s analysis of RSALs are

- Identifying bounding scenarios
- The numerical precision of the RSAL value
- Probability of exceeding a dose limit
- Robustness of our analysis of the RSAL
- The effect of time.

Criteria that have not been taken into account in *RAC*'s analysis of RSALs are

- Cost of cleanup
- Risks to the public during cleanup
- Ecological effect/impact of cleanup
- Risk as the basis for an RSAL
- Ambient plutonium levels in the environment
- Limits of detection of instrumentation
- Institutional controls
- Community values.

Some of these criteria are social, political, and economic factors that are outside the scope of our scientific work, yet their impact on the RSAL could be significant. We discuss each of these considerations below, listing those that we took into account first. Then we examine those that may impact the selection of an RSAL but were not included in our guidance.

10.1 Criteria That Have been Taken into Account in *RAC*'s Analysis of RSALs

10.1.1 Identifying Bounding Scenarios

Bounding scenarios are defined as exposure scenarios that describe probable lifestyles, behaviors, and land use that provide upper and lower limits on technically feasible RSALs. The bounding scenarios are developed in the RSAL analysis process as a means of (a) providing a plausible description of how individuals might use the land in the future, (b) allowing for the

uncertainty surrounding an unpredictable future, and (c) considering the protection of all members of the public at a dose level not to exceed 15 mrem y^{-1}. These bounding scenarios then provide a range of RSAL values from which to select the nominal value that best represents protection of public health. Inherent in all of our calculations of probability of exceeding the dose limit as a function of soil concentration is the impact of a prairie fire. This important consideration is critical in developing bounding scenarios.

10.1.2 The Numerical Precision of the RSAL Value

We recommend that RSAL values contain no more than two significant digits, which adequately represent the precision of the calculation. Assigning no more than two significant digits appropriately conveys the level of uncertainty associated with any methodology designed to derive soil action levels. Additionally, the RSAL should be rounded to the nearest multiple of 5 (e.g., 35 pCi g^{-1} instead of 33) when the RSAL is less than 100 and to 10 when the RSAL is greater than 100.

10.1.3 Probability of Exceeding a Dose Limit

Selecting a soil concentration from a probability figure, such as the one shown in Figure 9-1, necessarily implies selecting a probability level that conveys a degree of confidence that the soil concentration will not result in doses greater than the prescribed limit. Such a level will vary among individuals and reflect different social, political, economic, and scientific interests. *RAC* believes that a probability level of between 5% and 10% is reasonable for selecting a soil concentration to represent the RSAL.

This probability level is based on a number of things. First, CERCLA, the statute that applies in this case, indicates that the RSAL is intended to assure protection of the reasonable maximum exposed individual. The following quotes are typical of EPA guidance on this subject.

> ...actions at Superfund sites should be based on an estimate of the reasonable maximum exposure (RME) expected to occur under both current and future land use conditions. The reasonable maximum exposure is defined here as the highest exposure that is reasonably expected to occur at the site... (EPA 1988).

> The high-end of the risk distribution is, conceptually, above the 90th percentile of the actual (either measured of estimated) distribution. The conceptual range is not meant to precisely define the limits of this descriptor, but should be used carefully by the assessor as a target range for characterizing "high-end" risk. (Habicht 1992).

Additionally, ninety-percent confidence intervals are prevalent in practical parameter estimation, and the de facto default for tests of hypothesis is 5%, as almost any statistics text will indicate by its examples.

This considerable weight of evidence drew us to select the 5 to 10% level as an appropriate probability level. We recognize, however, that the probability level adopted by the RSALOP may be somewhat different and reflect other interests, values, and concerns.

10.1.4 Robustness of Our Analysis of the RSAL

The process of calculating RSALs is not an exact science. We rely on the current state of knowledge to make our calculations—knowledge that will certainly improve in the future. Because of inevitable changes and improvements in methods and data, some consideration should be given to developing an RSAL that is robust, or scientifically defensible with time and not likely to result in exceeding the 15 mrem y^{-1} target dose limit in the distant future. Although this concept cannot be accounted for directly, we believe that selecting a probability level that provides a reasonable margin for error (i.e., 5 to 10%) will help to address this criterion.

10.1.5 Effect of Time

RAC evaluated the time at which the maximum doses occur to ensure that the limiting scenario had been identified for the different radionuclides (see Appendix D). In all cases, the maximum doses occur at early times. It was found that doses are proportional to the amount of activity present in the surface soil compartment. Depletion of activity from this compartment is a function of the water infiltration rate and the soil-water distribution coefficient (K_d). Plutonium has a high value for K_d, which retards its movement from the surface soil and by the same token results in long transit times in the vadose zone. Consequently, during the time frame of interest and based on our analysis, plutonium does not reach the aquifer, and exposure via the groundwater pathway is zero. In contrast, uranium has a relatively low value for K_d, which results in higher leach rates and, therefore, more rapid depletion of uranium the surface soil. However, the dose that results from the groundwater pathway is smaller than from the surface soil pathways.

10.2 Criteria That Have Not Been Taken into Account in *RAC*'s Analysis of RSALs

10.2.1 Cost of Cleanup

The cost of cleanup was not considered in our selection of an RSAL. Weighing costs associated with achieving a soil action level must be carefully considered by the regulatory and funding agencies in conjunction with the stakeholders, site management, and, most importantly, those living near the site.

10.2.2 Ecological Effect/Impact of Cleanup

Any decision about a soil action level must take into account ecological impacts. A level that is too restrictive may severely affect the ecology of the site and may not be justified. This is a factor we did not consider in our analysis, but is important to consider in the future.

10.2.3 Risks to the Public during Cleanup

We have not considered the risks to the public from exposure to airborne and/or waterborne contaminants that may be possible during cleanup. It must be recognized that such risks may increase as the soil action level becomes more restrictive because more soil would be removed.

The remediation techniques used and attention that is given to limiting releases to the environment offsite during the remediation process must be considered. These risks have not been considered in our analysis.

10.2.4 Risk as the Basis for an RSAL

Although we examined the risks associated with the prescribed dose limit assuming the exposure is primarily from inhalation of plutonium (see Appendix E), we have not used risk as the basis for calculating an RSAL. The lifetime risk from plutonium exposure, assuming 70 years of exposure at 15 mrem y^{-1} was on the order of 5×10^{-4}, with a large uncertainty range from about 2×10^{-5} to 15×10^{-3}. The risk associated with exposure for 30 years at 15 mrem y^{-1} was approximately 2.3×10^{-4} with a range from 1×10^{-5} to 7×10^{-3}. This estimate was based on recent work in Grogan et al. (2000).

It is even more instructive to look at the range of risk associated with the possible exposures discussed in this report. At a minimum, a receptor in this analysis would be exposed to the maximum exposure from the fire in one year (15 mrem y^{-1}) and have no exposure in all other years. The maximum exposure would be 30 years at the dose maximum each year (15 mrem y^{-1}). The range established for exposures such as these would be from 7.5×10^{-6} to 2.3×10^{-4}. It is helpful to understand the risks associated with the dose limit.

10.2.5 Ambient Plutonium Levels in the Environment

Plutonium in Colorado soil resulting from fallout from nuclear weapons tests has not been taken into account in our methodology. Although ambient plutonium in the Rocky Flats area from sources not attributable to Rocky Flats is higher than that seen in other parts of the country (see Appendix C), the background levels in soil (0.008–0.1 pCi g^{-1}) are still much smaller than what will ultimately be used for a soil action level. It is important to consider this ambient level of plutonium in the environment because it helps put into perspective any value of action level that is selected.

10.2.6 Limits of Detection of Instrumentation

The ability to detect plutonium contamination at a certain level could influence a decision about the RSAL. If the value is below a limit of detection, it is not possible to verify the cleanup level being implemented. The detection limit of plutonium should be considered in the final selection of a soil action level but was not taken into account in our analysis.

10.2.7 Institutional Controls

The decision about whether or not institutional controls will continue to exist at the RFETS facility is a political one and was not taken into account in our work. Presumably, these controls could significantly impact the economics of cleanup and the long-term future use of the site. Decisions regarding institutional controls must remain the responsibility of stakeholders and parties responsible for the site.

10.2.8 Community Values

In selecting an RSAL, one factor stands out as possibly the most important for all stakeholders to consider. In a sense, it is scientific in that it is based on a level of calculated risk or dose. On the other hand, it is a subjective decision that may well be different for every individual. The community must weigh the ethics and values inherent in a decision regarding a contaminated environment that will eventually be regarded as acceptable for individuals to inhabit. This consideration has not been addressed in our analysis, but it is one that must ultimately be taken into account.

11. RESULTS

11.1 Introduction to Plutonium RSALS

The conceptual site model *RAC* used to calculate plutonium RSALs was based on a heterogeneous distribution of plutonium soil and air concentrations across the RFETS. An important distinction between this method and the one used by DOE/EPA/CDPHE (1996) and inherent to the RESRAD model is that the conceptual model employed in RESRAD assumes soil and air contamination to be homogeneous (uniformly distributed) across the site being modeled. When soil and air concentrations are assumed to be homogeneous, the dose to a receptor is proportional to the soil concentration in the area being modeled and the calculation of an RSAL is straightforward. Because a homogeneous distribution of plutonium is definitely not the case at the RFETS, we used the air dispersion model described earlier to incorporate soil and air concentration heterogeneity into the calculation. We used RESRAD only to calculate intakes and doses.

Incorporating soil and air concentration heterogeneity into the conceptual model complicates both the calculation and interpretation of RSALs. A conventionally defined RSAL (such as defined in RESRAD) assumes homogeneous concentration and a fixed dose-to-source ratio (the source in this case is the contaminated soil). In our calculation, the RSAL depends not only on the receptor scenario parameters but also on the location of the receptor relative to sources of contamination. Consequently, different RSALs are calculated for the same scenario depending on where the receptor is located. Furthermore, the conventional RSAL as defined in RESRAD is not really applicable to the conceptual model we employed in our calculations because it is based on a uniformly contaminated area, which is not true of the RFETS soil concentration profile.

The RSALs we calculated are a combination of both near-field and far-field contamination sources. The near-field source is the soil contamination at the location of the receptor. The far-field source is resuspended contaminated soil from upwind sources. The relevant pathways of exposure that are affected by the far-field source include inhalation, plant ingestion (from foliar deposition), and milk and meat ingestion (again from foliar deposition).

Our research indicates that the *fraction* of total dose from the far-field pathways is nearly proportional to the air-to-soil concentration ratio, which varies with location across the site. The air-to-soil concentration ratio is the calibrated air concentration at a given location divided by the soil concentration at that same location. The air-to-soil concentration ratio can be equated to the mass loading factor in RESRAD. If we use the air dispersion model to calculate an air-to-soil concentration ratio representative of worst case conditions at the site, then we can calculate RSALs in a manner similar to RESRAD, with the understanding that these values will likely error on the side of conservatism.

It is important to remember that the actual soil concentration does not factor into the calculation of an RSAL. The RSAL is only a function of the dose-to-source ratio; that is, the dose per unit concentration in soil. The dose-to-source ratio is a function of the physical transport processes and assumed receptor behavior (i.e., ingestion and inhalation rates). In our model, the dose-to-source ratio varies across the site because the air-to-soil concentration ratio varies. Because our objective was to provide a conservative RSAL, we selected a location with the highest dose-to-source ratio. This location corresponds to the location with the highest air-to-soil concentration ratio.

Therefore, to provide RSALs that could be applied across the RFETS, we located each receptor at the point of the maximum air-to-soil concentration ratio, which was the east edge of the site near Indiana Street. This location had a geometric mean soil concentration of 1.81 pCi g^{-1} and an air concentration of 2.33×10^{-5} pCi m^{-3}, which yields an air-to-soil concentration ratio of 1.29×10^{-5} g m^{-3}. Air concentrations at this location are proportionally higher relative to the soil concentration because the air concentrations reflect the cumulative flux from all upwind-contaminated areas.

The RSALs that result from this methodology may be interpreted as a clean-up level applied to the entire site. That is, contaminated areas, as defined by the soil sampling protocol, should not exceed the RSAL. It is important to note that *doses* at the location where these RSALs are calculated (near Indiana Street) did not exceed the 15 mrem dose limit (at the 90th percentile level) based on current contamination levels. The RSAL, however, is more an artifact of the calculation than a statement of the actual heath risk a person may incur at that location. We believe these values do provide an estimate of soil concentrations that are protective of human health for the RFETS. However, it should be understood that the RSALs calculated in this manner are not necessarily consistent with the conceptual model of heterogeneous contamination developed for this study. Section 7 of this report presented an alternative method for calculating an RSAL that used the heterogeneous features of our conceptual model. This method was based on the dose a receptor receives at a given location and a proposed level of remediation and provided us with a confirmation of the viability of the RSAL results presented in Section 11.4.

11.2 Important Pathways for Plutonium Isotope Simulations

Figure 11-1 illustrates the relative importance of pathways for plutonium RSAL simulations. This figure shows the fraction of the total dose for the three most important exposure pathways as a function of the $^{239+240}$Pu RSAL for the rancher scenario. The $^{239+240}$Pu RSAL includes the sum-of-ratios calculation described in Equation (5-5). There is an inverse relationship between the fraction of dose attributed to inhalation and the $^{239+240}$Pu RSAL because the lower RSALs are driven by the occurrence of a fire. The occurrence of a fire results in enhanced resuspension and, therefore, higher air concentrations for the same soil concentration, which leads to higher inhalation doses. As the importance of inhalation decreases, the importance of the other pathways increases. Most notably, soil ingestion becomes the most important exposure pathway. The groundwater pathway was negligible in all cases. RSALs were the same with and without the pathway turned on. Therefore, RSAL results are only shown without the groundwater pathway turned on for the plutonium isotopes.

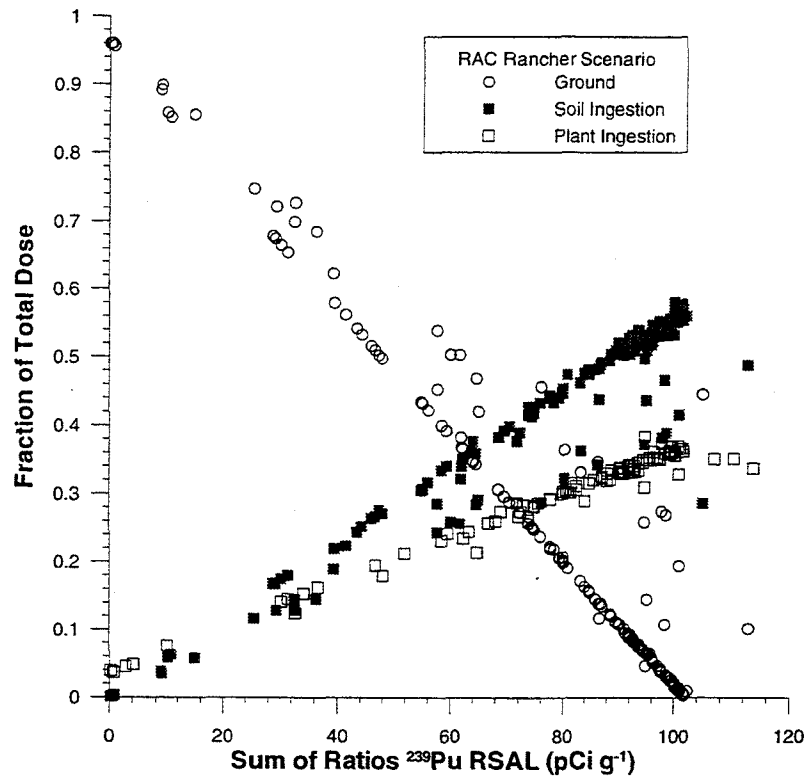

Figure 11-1. Fraction of the total dose for the rancher scenario as a function of the $^{239+240}Pu$ sum-of-ratios RSAL for the three primary exposure pathways (inhalation, soil ingestion, and plant ingestion).

11.3 Introduction to Uranium RSALS

Uranium was treated differently than plutonium because uranium contamination around the RFETS is more localized and covers a smaller area, as discussed in the uranium methodology section. We treated uranium contamination as an isolated hot spot problem and assumed a uniformly contaminated area of 100 m^2. Under these conditions, the RESRAD conceptual model is applicable with no modifications and calculation of an RSAL is straightforward.

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11.4 Scenario Probability Curves for Plutonium

11.4.1 Scenario DOE-1

Scenario proposed by: DOE/EPA/CDPHE **Scenario name:** Resident

Some key scenario parameters:

Dose limit (mrem y^{-1})	15 and 85
Time on the site (h y^{-1})	8400
Time indoors onsite (%)	100
Breathing rate (m^3 y^{-1})	7000
Soil ingestion (g y^{-1})	70
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	no

RSAL probability curve resulting from *RAC* calculations:

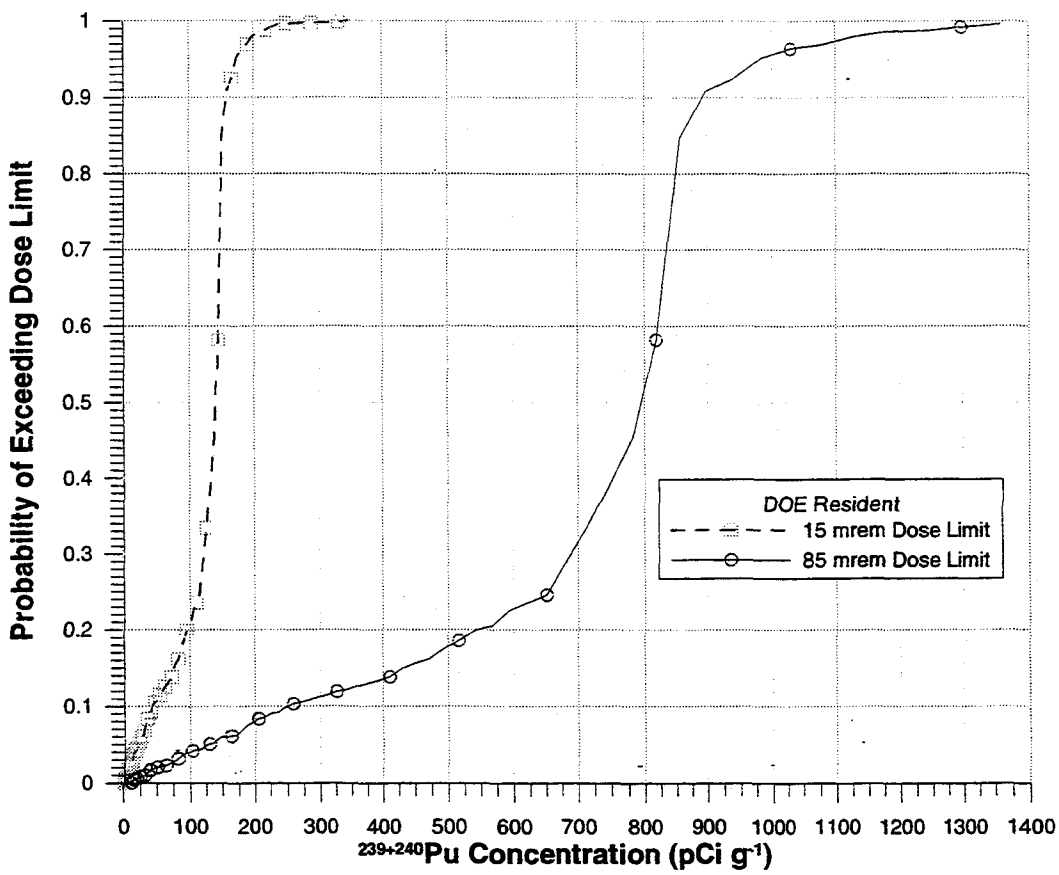

Figure 11-2. Curve representing the probability of exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996). The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSALs for $^{239+240}$Pu (including the sum-of-ratios calculation) were 259 pCi g^{-1} for the 85 mrem dose limit and 45 pCi g^{-1} at the 15 mrem dose limit. Individual RSALs for the plutonium isotopes are presented in Table 11-1.

Table 11-1. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the DOE/EPA/CDPHE Resident Scenario using *RAC* stochastic methodology, 15 mrem dose limit (pCi g^{-1})[a]

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	15	42	6	7	58	7
5%	45	116	27	29	175	28
10%	89	151	54	55	330	56
50%	205	205	176	178	900	184
90%	552	17650	192	192	2055	201
95%	917	91820	200	205	3359	208

[a]The individual RSALs for the 85 mrem dose limit can be obtained by multiplying these values by $^{85}/_{15}$ or about 5.67.

This section illustrates the RSALs using the resident scenario from DOE/EPA/CDPHE (1996) and the *RAC* methodology presented in this report. The pathways of concern were different because the methodologies were different and a number of the input parameters were changed in our calculation.

11.4.2 Scenario DOE-2

Scenario proposed by: DOE/EPA/CDPHE **Scenario name:** Open space user

Some key scenario parameters:

Dose limit (mrem y^{-1})	85
Time on the site (h y^{-1})	125
Time outdoors onsite (%)	100
Breathing rate (m^3 y^{-1})	175
Soil ingestion (g y^{-1})	2.5
Irrigation water source	not applicable
Irrigation rate (m y^{-1})	not applicable
Onsite drinking water source	no

RSAL probability curve resulting from *RAC* calculations:

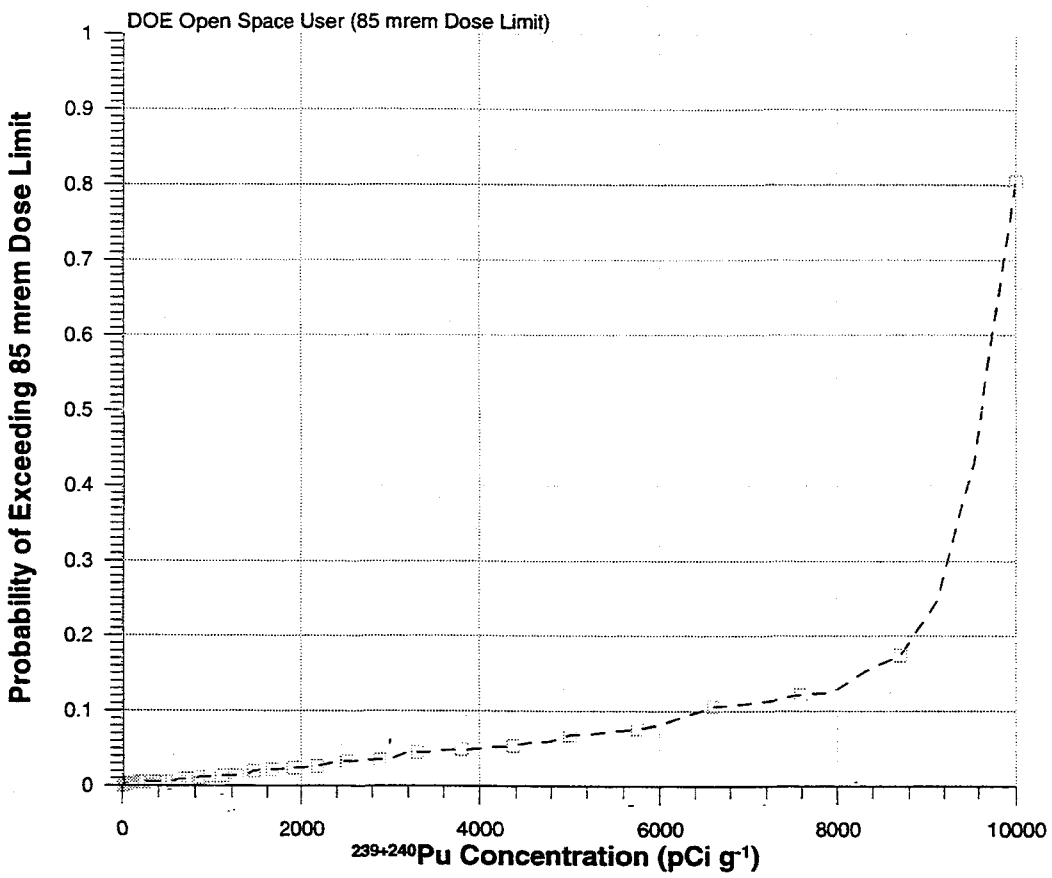

Figure 11-3. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE Open space user scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996) and assumed that the site remains as open space and will not be developed in the future. The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSAL for $^{239+240}$Pu (including the sum-of-ratios calculation) was 6600 pCi g^{-1} for an 85 mrem dose limit. Individual RSALs for the plutonium isotopes are presented in Table 11-2.

Table 11-2. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the DOE/EPA/CDPHE Open Space User, 85 mrem dose limit (pCi g^{-1})

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	1882	9732	934	990	6990	969
5%	3549	24900	5790	6044	14840	6040
10%	3710	30200	11510	12180	17070	12140
50%	3820	36600	30300	32200	18600	33400
90%	11300	3985000	32250	34200	41400	35800
95%	15900	11820000	32400	34300	58360	35930

This section illustrates the RSALs using the open space user scenario from DOE/EPA/CDPHE (1996) and the *RAC* methodology presented in this report. The pathways of concern were different because the methodologies were different and a number of the input parameters were changed in our calculation.

11.4.3 Scenario DOE-3

Scenario proposed by: DOE/EPA/CDPHE **Scenario name:** Office worker

Some key scenario parameters:

Dose limit (mrem y^{-1})	85
Time on the site (h y^{-1})	2000
Time indoors onsite (%)	100
Breathing rate (m^3 y^{-1})	1660
Soil ingestion (g y^{-1})	12.5
Irrigation water source	not applicable
Irrigation rate (m y^{-1})	not applicable
Onsite drinking water source	no

RSAL probability curve resulting from *RAC* calculations:

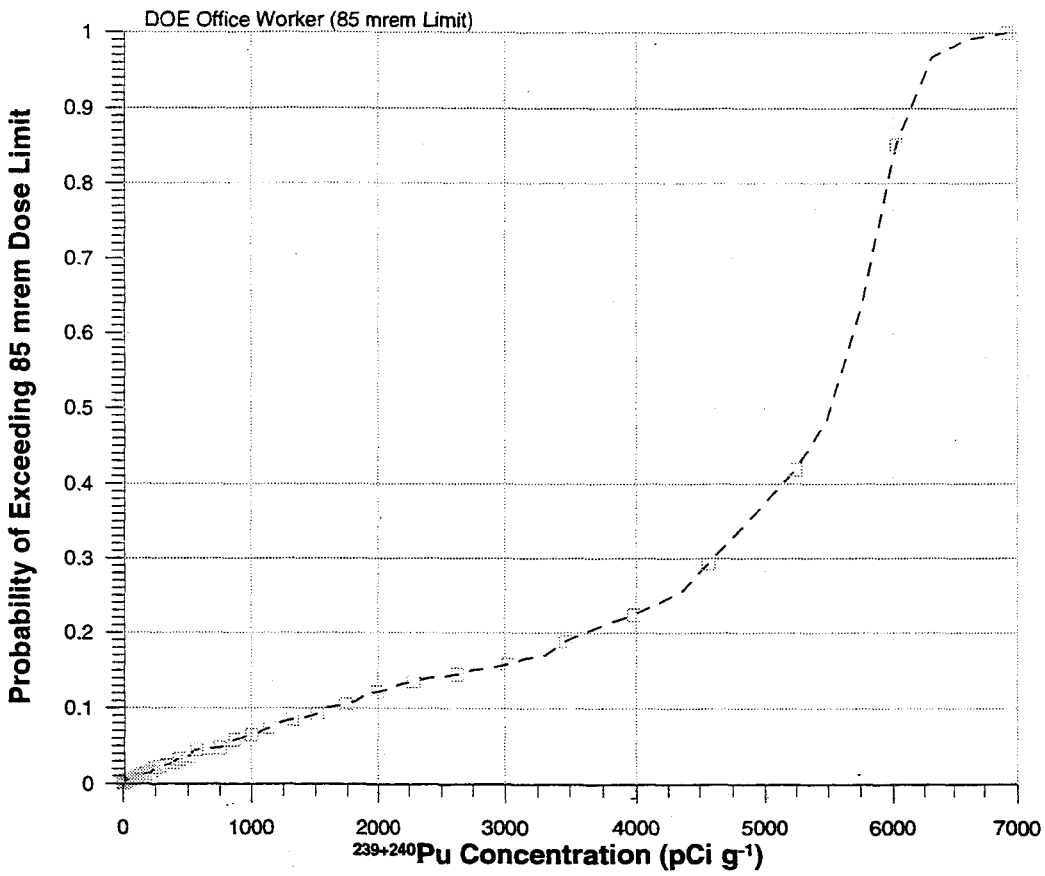

Figure 11-4. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE office worker scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996) and assumes that the site is developed into an industrial park/office complex. The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the *RAC*-calculated RSAL for $^{239+240}$Pu (including the sum-of-ratios calculation) was 1585 pCi g^{-1} for an 85 mrem dose limit. Individual RSALs for the plutonium isotopes are presented in Table 11-3.

Table 11-3. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the DOE/EPA/CDPHE Office Worker Scenario, 85 mrem dose limit (pCi g^{-1})

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	339	1587	142	151	1244	147
5%	1679	4664	924	956	6462	954
10%	3517	5950	1948	2025	12870	2011
50%	10800	7750	6540	6600	45500	6840
90%	18150	716000	7190	7220	66600	7530
95%	25260	3140000	7230	7260	92860	7570

This section illustrates the RSALs using the office worker scenario from DOE/EPA/CDPHE (1996) and the *RAC* methodology presented in this report. The pathways of concern were different because the methodologies were different and a number of the input parameters were changed in our calculation.

11.4.4 Scenario *RAC*-1

Scenario proposed by: *RAC* **Scenario name:** Resident rancher

Some key scenario parameters:

Dose limit (mrem y^{-1})	15
Time on the site (h y^{-1})	8760
Time indoors onsite (%)	60
Breathing rate (m^3 y^{-1})	10800
Soil ingestion (g y^{-1})	75
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	groundwater

RSAL probability curve resulting from *RAC* calculations:

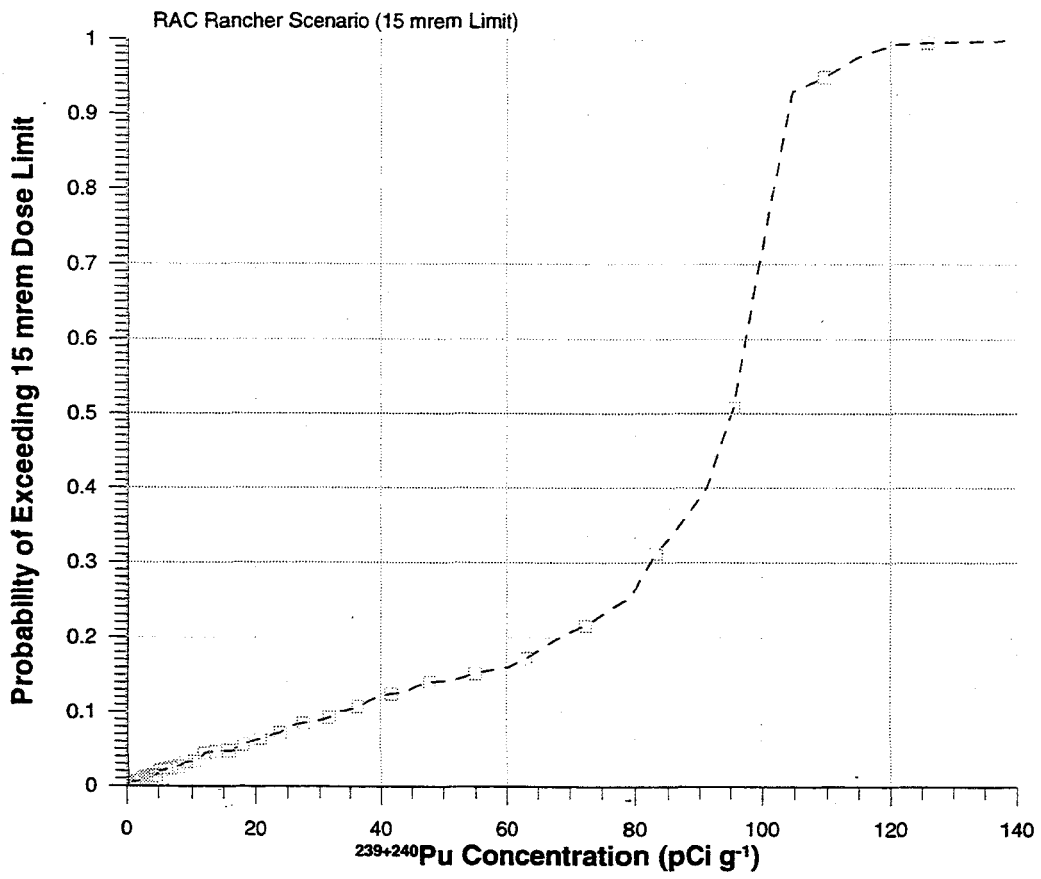

Figure 11-5. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario represents a full-time adult rancher who lives and works on RFETS lands. The probability curve shows two distinct slopes (Figure 11-5). For $^{239+240}Pu$ concentrations less than ~80 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants. For soil concentrations greater than 80 pCi g^{-1}, the slope of the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The steep slope of the probability curve for $^{239+240}Pu$ concentrations greater than ~80 pCi g^{-1} results from less variability in the doses from the soil and plant ingestion pathways compared to the inhalation pathway. Inhalation doses were proportional to the estimated air concentration, and air concentrations were considerably more variable than soil concentrations. Therefore, RSALs at the 10% probability level (90% probability that the 15 mrem dose limit will *not* be exceeded) were controlled mainly by the inhalation of resuspended dust. Note that the characteristic inflection point of this probability curve is also seen in the probability curves for the other exposure scenarios. At the 10% probability level, the $^{239+240}Pu$ RSAL (including the sum of ratios calculation) was 33 pCi g^{-1}. Individual plutonium isotope RSALs are given in Table 11-4.

Table 11-4. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the *RAC* Rancher Scenario, 15 mrem dose limit (pCi g^{-1})

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	8	34	3	3	28	3
5%	35	92	20	21	132	21
10%	66	114	41	42	242	42
50%	128	142	121	122	569	127
90%	236	15150	131	132	863	137
95%	389	60330	132	132	1426	138

11.4.5 Scenario *RAC*-2

Scenario proposed by: *RAC* **Scenario name:** Child of rancher (10 years old)

Some key scenario parameters:

Dose limit (mrem y^{-1})	15
Time on the site (h y^{-1})	8760
Time indoors onsite (%)	75
Breathing rate (m^3 y^{-1})	8600
Soil ingestion (g y^{-1})	75
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	groundwater

RSAL probability curve resulting from *RAC* calculations:

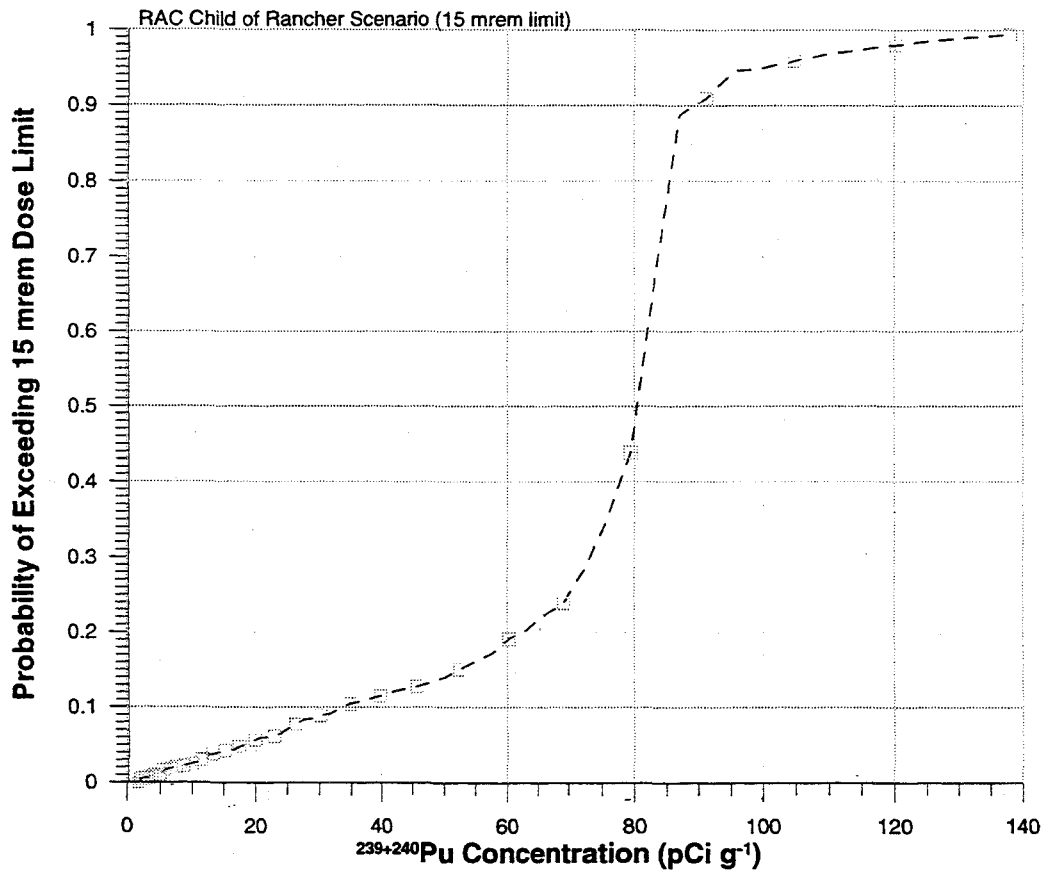

Figure 11-6. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario represents a 10-year old child of a full time resident (rancher) who lives on RFETS lands. The probability curve shows two distinct slopes (Figure 11-6). For $^{239+240}Pu$ concentrations less than ~60 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants, primarily from fire events. For soil concentrations greater than 60 pCi g^{-1}, the slope of the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The inflection point of this probability curve occurs at a lower $^{239+240}Pu$ soil concentration compared to the adult rancher. Because ingestion rates for the two scenarios were assumed to be the same (75 g y^{-1}), this difference reflects the differences in the ingestion dose conversion factors between the adult and child. At the 10% probability level, the $^{239+240}Pu$ RSAL (including the sum-of-ratios calculation) was 35 pCi g^{-1}. Individual plutonium isotope RSALs are given in Table 11-5.

Table 11-5. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the *RAC* Child of Rancher Scenario, 15 mrem dose limit (pCi g^{-1})

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	15	31	5	5	58	6
5%	46	75	21	22	173	23
10%	89	90	39	39	330	41
50%	198	108	93	93	816	97
90%	537	11150	98	98	2010	102
95%	898	59000	109	114	3290	114

11.4.6 Scenario *RAC*-3

Scenario proposed by: *RAC* **Scenario name:** Infant of rancher (2 years old)

Some key scenario parameters:

Dose limit (mrem y^{-1})	15
Time on the site (h y^{-1})	8760
Time indoors onsite (%)	90
Breathing rate (m^3 y^{-1})	1900
Soil ingestion (g y^{-1})	75
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	groundwater

RSAL probability curve resulting from *RAC* calculations:

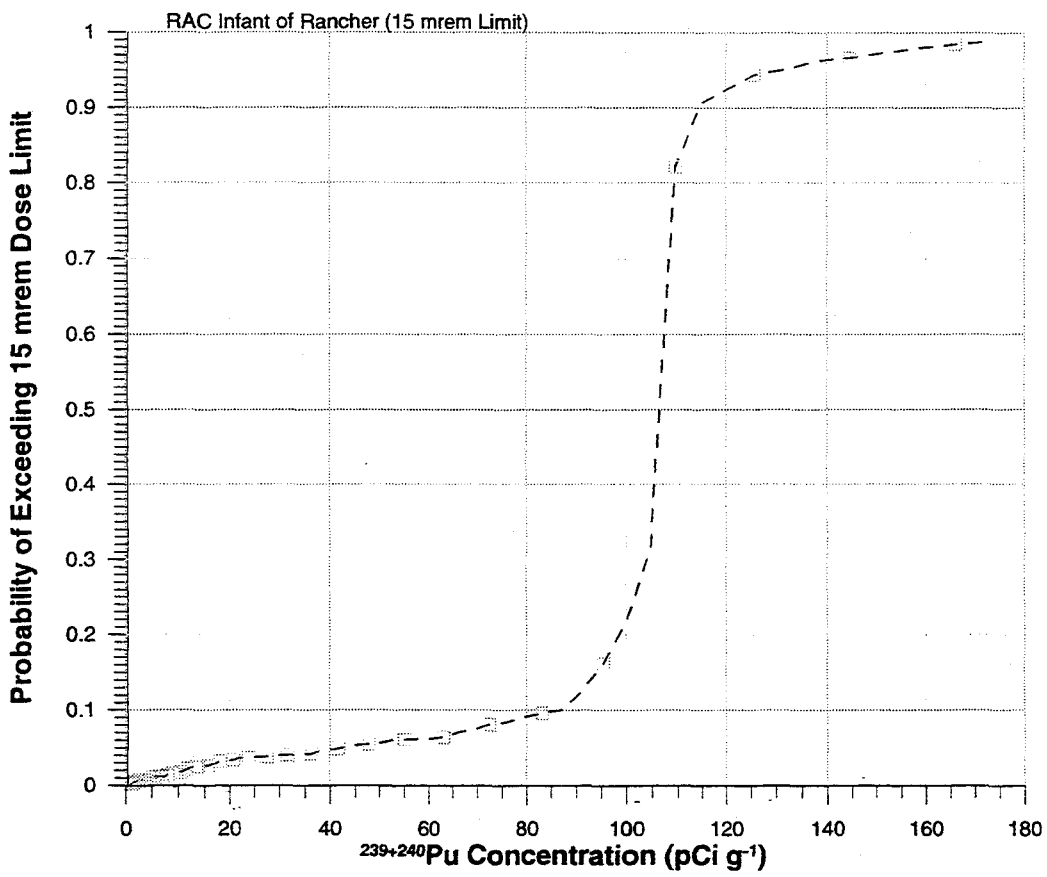

Figure 11-7. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* infant of rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario represents an infant of a full time resident (rancher) who lives on RFETS lands. Like the other scenarios, the probability curve shows two distinct slopes (Figure 11-7). For $^{239+240}Pu$ concentrations less than ~90 pCi g^{-1}, the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust and foliar deposition on plants. For soil concentrations greater than 90 pCi g^{-1}, the slope of the probability curve exhibits a steeper slope and is controlled mainly by the soil ingestion and plant ingestion pathways. The inflection point of this probability curve occurs at a higher $^{239+240}Pu$ soil concentration compared to the adult rancher and child scenarios. This difference reflects the differences in the dose conversion factors and intake rates of contaminated media for the adult, child and infant. While the dose conversion factors are generally higher for infants, their contaminant intake rates (i.e., breathing rate and food ingestion rates) are generally lower. At the 10% probability level, the $^{239+240}Pu$ RSAL (including the sum-of-ratios calculation) was 87 pCi g^{-1}. Individual plutonium isotope RSALs are given in Table 11-6.

Table 11-6. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the *RAC* Infant of Rancher Scenario, 15 mrem dose limit (pCi g^{-1})

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	13	45	6	6	46	6
5%	113	112	49	50	443	52
10%	182	120	95	99	792	99
50%	233	133	126	126	1020	132
90%	398	5400	129	129	1550	135
95%	677	54910	142	148	2483	148

11.4.7 Scenario *RAC*-4

Scenario proposed by: *RAC* **Scenario name:** Current site industrial worker

Some key scenario parameters:

Dose limit (mrem y^{-1})	85 and 15
Time on the site (h y^{-1})	2100
Time indoors onsite (%)	40
Breathing rate (m^3 y^{-1})	3700
Soil ingestion (g y^{-1})	50
Irrigation water source	not applicable
Irrigation rate (m y^{-1})	not applicable
Onsite drinking water source	no

RSAL probability curve resulting from *RAC* calculations:

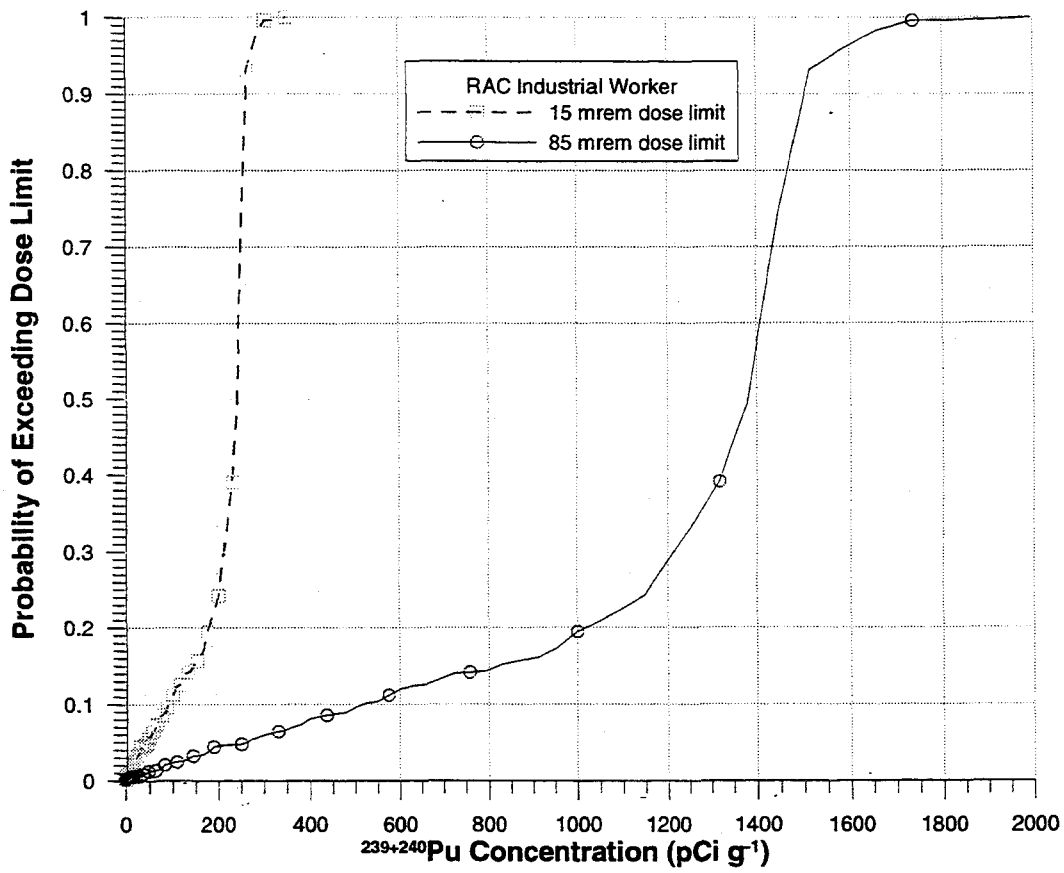

Figure 11-8. Probability of the total dose exceeding the 15 and 85 mrem dose limits for the *RAC* site industrial worker scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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This scenario represents an adult who works at an industrial complex at the RFETS. Like the other scenarios, the probability curve shows two distinct slopes (Figure 11-8). For $^{239+240}Pu$ concentrations less than ~150 pCi g^{-1} (~850 pCi g^{-1} for the 85 mrem dose limit), the slope of the probability curve is shallow and reflects doses from inhalation of resuspended dust (plant ingestion was not considered). For soil concentrations greater than 150 pCi g^{-1} (850 pCi g^{-1} for the 85 mrem dose limit), the slope of the probability curve exhibits a steeper slope and is controlled by soil ingestion. The inflection point of this probability curve occurs at a higher $^{239+240}Pu$ soil concentration compared to all other *RAC* scenarios because intake rates of contaminated media are substantially less for this scenario. At the 10% probability level, the $^{239+240}Pu$ RSALs (including the sum-of-ratios calculation) were 92 pCi g^{-1} at the 15 mrem dose limit and 525 pCi g^{-1} at the 85 mrem dose limit. Individual plutonium isotope RSALs for an 85 mrem dose limit are given in Table 11-7.

Table 11-7. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the *RAC* Site Industrial Worker Scenario, 85 mrem dose limit (pCi g^{-1})[a]

Percentile	Am-241	Pu-238	Pu-239	Pu-240	Pu-241	Pu-242
1%	120	522	50	53	441	52
5%	560	1330	313	324	2105	323
10%	1077	1617	626	650	3948	646
50%	2250	1950	1690	1710	9730	1770
90%	4080	230000	1800	1810	15000	1890
95%	6527	854800	1810	1820	23890	1900

[a]The individual RSALs for the 15 mrem dose limit can be obtained by multiplying the values in this table by $^{15}/_{85}$ or about 0.18

11.5 Scenario probability curves for Uranium

Radionuclide soil action levels are presented for uranium isotopes (^{234}U, ^{235}U, and ^{238}U) for three scenarios: the DOE/EPA/CDPHE resident, the *RAC* rancher, and *RAC* child scenarios. The DOE/EPA/CDPHE resident scenario was chosen for comparison between our methodology and that of DOE/EPA/CDPHE. The rancher and child scenarios were chosen because they resulted in the most restrictive RSALs for plutonium. The prairie fire was considered for the uranium but was found to be of little consequence because doses were dominated by groundwater, plant and soil ingestion, and ground exposure.

11.5.1 Scenario DOE-1

Scenario proposed by: DOE/EPA/CDPHE **Scenario name:** Resident

Some key scenario parameters:

Dose limit (mrem y^{-1})	15 and 85
Time on the site (h y^{-1})	8400
Time indoors onsite (%)	100
Breathing rate (m^3 y^{-1})	7000
Soil ingestion (g y^{-1})	70
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	no

RSAL probability curve resulting from *RAC* calculations:

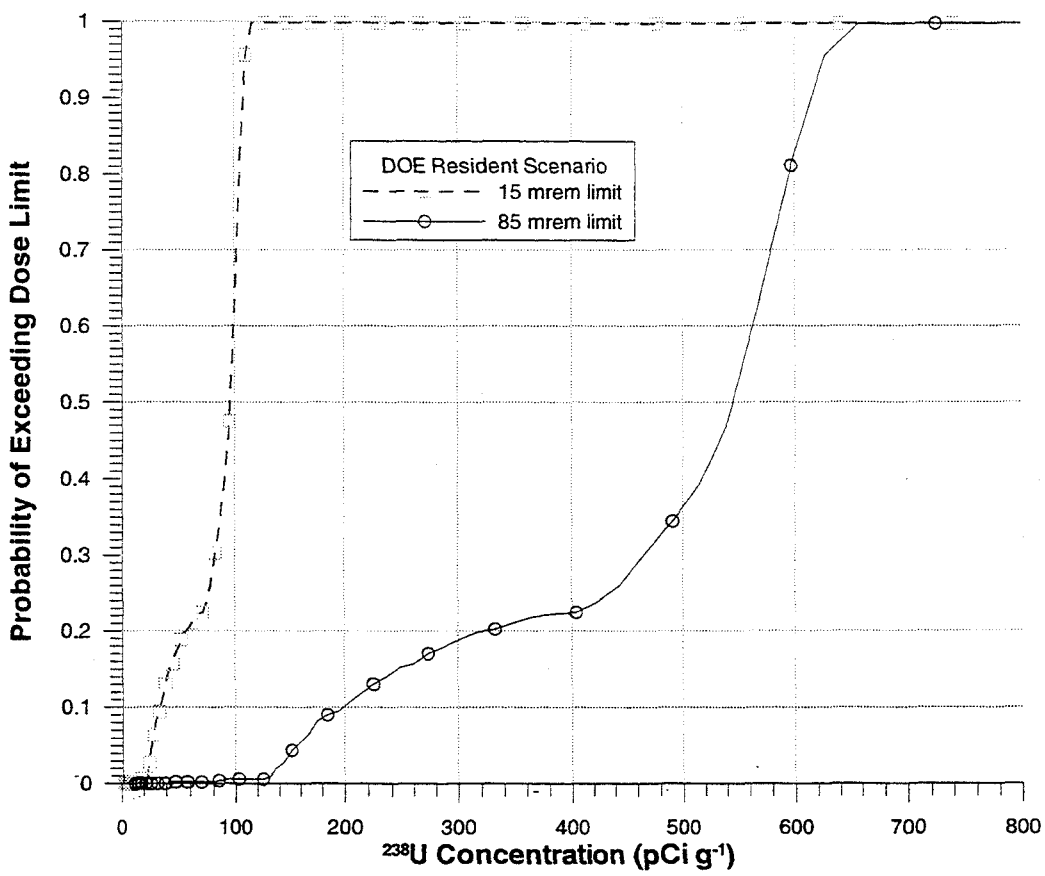

Figure 11-9. Probability of the total dose exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario. Total dose includes all uranium isotopes and their daughter products.

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This scenario was part of the original RSAL calculation (DOE/EPA/CDPHE 1996). The RSALs presented here represent this same scenario calculated stochastically using the methodology developed by *RAC*. At the 10% level (90% probability that the dose limit would *not* be exceeded), the ^{238}U RSALs were ~35 pCi g^{-1} for the 15 mrem dose limit and ~200 pCi g^{-1} at the 85 mrem dose limit (Figure 11-9). These RSALs incorporated the sum-of-ratios calculation to include the other uranium isotopes. Percentiles of the individual RSALs without the sum-of-ratios calculation are presented in Table 11-8.

Table 11-8. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the DOE/EPA/CDPHE Resident Scenario (pCi g^{-1})

	85 mrem dose limit			15 mrem dose limit		
Percentile	U-234	U-235	U-238	U-234	U-235	U-238
1%	263	137	284	46	24	50
5%	305	137	330	54	24	58
10%	394	137	425	69	24	75
50%	7242	137	751	1278	24	133
90%	7322	543	753	1292	96	133
95%	7336	712	759	1295	126	134

A significant difference between the DOE/EPA/CDPHE methodology and our methodology was in the area assigned to uranium contamination. The DOE/EPA/CDPHE methodology assumed the area of uranium contamination was the same as plutonium (40,000 m^2). Our investigation indicated that uranium contamination was not as widespread as plutonium and was mainly limited to past disposal areas or burn pits. We, therefore, treated the uranium contamination as a hot spot and restricted its area to 100 m^2. As a result, the inhalation pathway was less important in our simulations than those of DOE/EPA/CDPHE.

Probably of greater importance was the way the groundwater pathway was treated between the *RAC* and DOE/EPA/CDPHE interpretations of this scenario. DOE/EPA/CDPHE did not account for the groundwater pathway and extracted doses for the year 2000. However, they allowed uranium to be leached from the ground surface at a rate proportional to the background infiltration rate (0.38 m y^{-1}) plus the irrigation rate (1 m y^{-1}). In our calculations, we let RESRAD calculate the maximum dose in the 1000-year time of compliance and extracted RSALs for that time. The time of maximum dose varied between years 2000 and 2500 depending on the contaminant travel times in the unsaturated and saturated zone. Uranium that migrated to the groundwater was then used for irrigation, thereby contaminating edible plants (direct consumption of water was not considered).

This section illustrates the RSALs using the resident scenario from DOE/EPA/CDPHE (1996) and the *RAC* methodology presented in this report. The pathways of concern are different because we included an evaluation of the groundwater pathway and a number of the input parameters have changed in our calculation.

11.5.2 Scenario *RAC*-1

Scenario proposed by: *RAC* **Scenario name:** Resident rancher

Some key scenario parameters:

Dose limit (mrem y^{-1})	15
Time on the site (h y^{-1})	8760
Time indoors onsite (%)	60
Breathing rate (m^3 y^{-1})	10800
Soil ingestion (g y^{-1})	75
Irrigation water source	groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	groundwater

RSAL probability curve resulting from *RAC* calculations:

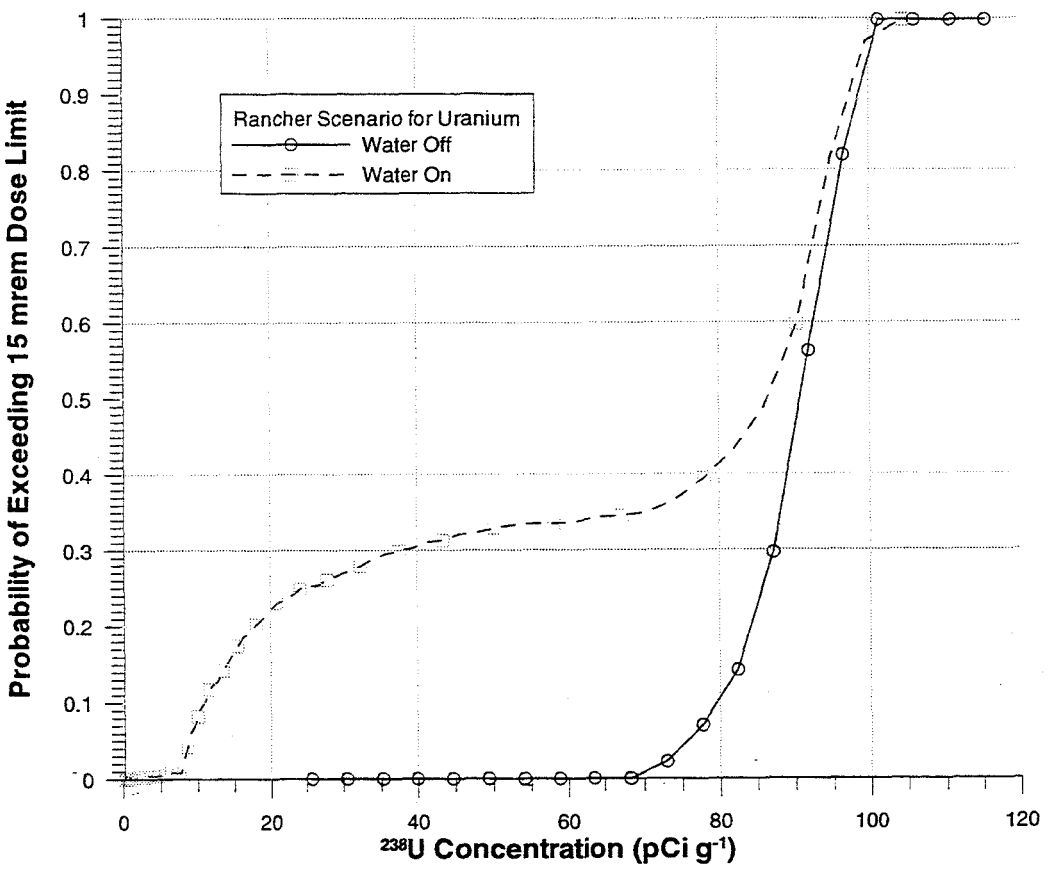

Figure 11-10. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario. Total dose includes all uranium isotopes and their daughter products.

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Soil action levels were calculated for two cases: one that considered a viable groundwater pathway and the other that assumed all water was derived from offsite sources. Differences between the RSALs with the water pathway on and off were substantial. When the water pathways are turned on, a 1 m y^{-1} irrigation rate was used and resulted in a substantial increase in the removal of radionuclides from surface soil via leaching. However, unlike plutonium, unsaturated zone transit times (the time it takes radionuclides to travel from the contaminated zone to the aquifer) were typically were less than 500 years for uranium isotopes. Consequently, the dose as a function of time typically had two peaks: one at year 2000 (the start time of the simulation) and one after uranium reached the water well in the aquifer. At the 10% level (a 90% probability that the 15 mrem dose limit will *not* be exceeded), the RSAL for ^{238}U (including the sum-of-ratios calculation) with the water pathway on was 11 pCi g^{-1}, and the RSAL with the water pathway off was 80 pCi g^{-1} (Figure 11-10). Percentiles of the individual RSALs without the sum-of-ratios calculation are presented in Table 11-9. Note that for the water pathway off, the RSALs show little variance and are almost identical (with rounding) for all percentiles. The sum-of-ratios calculations incorporated uncertainty in the isotopic ratios, and that is reflected in the water off curve in Figure 11-10.

As shown in Figures 6-1 and 6-2 in the uranium methodology section, doses were dominated by water dependent pathways for ^{238}U RSALs <60 pCi g^{-1} with the water pathway on. With the water pathway off, doses were driven by ground exposure and plant ingestion.

Table 11-9. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the *RAC* Resident Rancher Scenario (pCi g^{-1})

	Water on			Water off		
Percentile	U-234	U-235	U-238	U-234	U-235	U-238
1%	16	11	17	492	2	9
5%	18	18	19	493	28	134
10%	21	22	23	494	28	134
50%	501	28	134	497	28	134
90%	501	51	134	499	28	134
95%	501	79	134	499	28	134

11.5.3 Scenario *RAC*-2

Scenario proposed by: *RAC* **Scenario name:** Child of rancher (10 years old)

Some key scenario parameters:

Dose limit (mrem y^{-1})	15
Time on the site (h y^{-1})	8760
Time indoors onsite (%)	90
Breathing rate (m^3 y^{-1})	8600
Soil ingestion (g y^{-1})	75
Irrigation water source	Groundwater
Irrigation rate (m y^{-1})	1
Onsite drinking water source	Groundwater

RSAL probability curve resulting from *RAC* calculations:

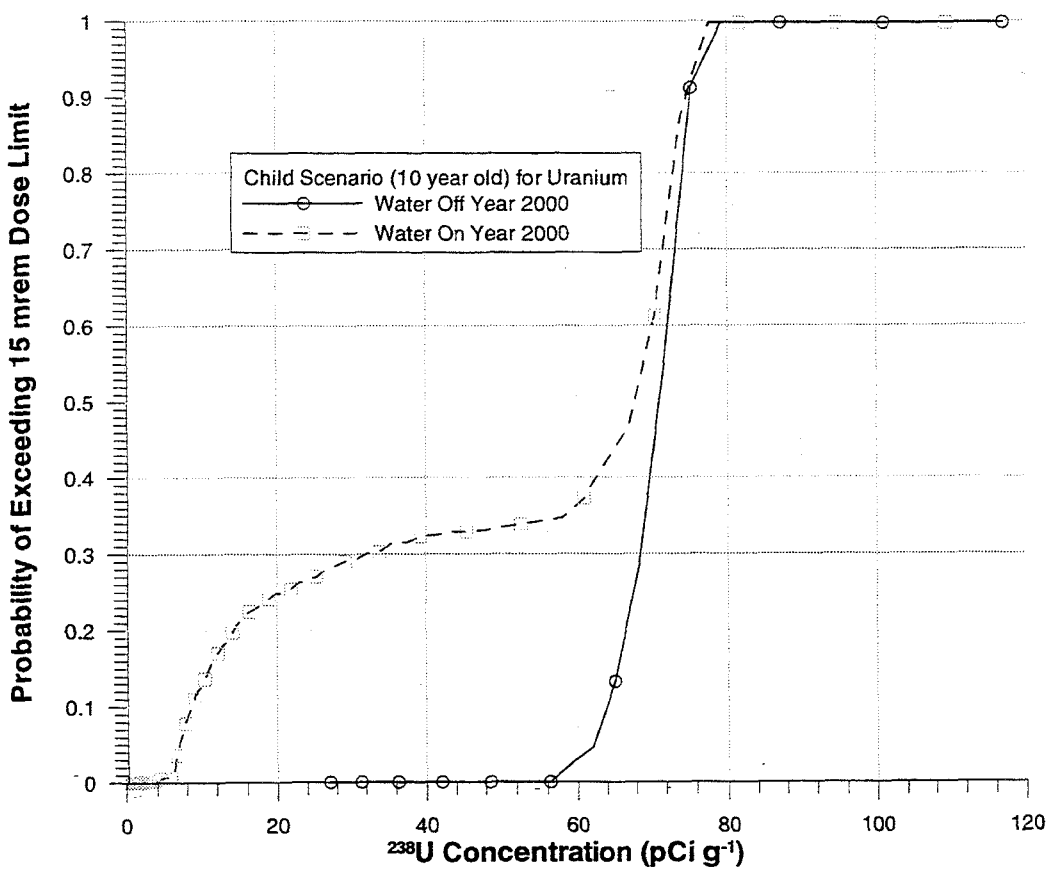

Figure 11-11. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario. Total dose includes all uranium isotopes and their daughter products.

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Soil action levels were calculated for two cases: one that considered a viable groundwater pathway and the other that assumed all water was derived from offsite sources. Differences between the RSALs with the water pathway on and off were substantial. When the water pathways are turned on, a 1 m y^{-1} irrigation rate was applied to the soil surface and resulted in a substantial depletion of radionuclides from surface soil via leaching. However, unlike plutonium, unsaturated zone transit times (the time it takes radionuclides to travel from the contaminated zone to the aquifer) were typically were less than 500 years for uranium isotopes. Consequently, the dose as a function of time typically had two peaks: one at year 2000 (the start time of the simulation) and one after uranium reached the water well in the aquifer. At the 10% level (a 90% probability that the 15 mrem dose limit will *not* be exceeded), the RSAL for ^{238}U (including the sum-of-ratios calculation) with the water pathway on was 9 pCi g^{-1} and the RSAL with the water pathway off was 65 pCi g^{-1}. Percentiles of the individual RSALs without the sum-of-ratios calculation are presented in Table 11-10.

Table 11-10. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the *RAC* Child of Rancher Scenario (pCi g^{-1})

	Water On			Water Off		
Percentile	U-234	U-235	U-238	U-234	U-235	U-238
1%	12	13	13	276	27	111
5%	14	14	15	276	27	112
10%	17	18	19	276	27	112
50%	277	27	112	277	27	112
90%	278	43	112	278	27	112
95%	278	63	112	278	27	112

As shown in Figures 6-1 and 6-2 in the uranium methodology section, doses were dominated by water dependent pathways for ^{238}U RSALs <60 pCi g^{-1} with the water pathway on. With the water pathway off, doses were driven by ground exposure and plant ingestion.

12. DISCUSSION AND RECOMMENDATIONS FOR FURTHER RESEARCH

Objectives of this project were a review of agency proposals for Rocky Flats site remediation and an independent calculation of radionuclide soil action levels for the site. In addition to meeting the contractual requirements, we have developed methods to answer the questions implied not only by applicable regulatory limits but also by the environmental models and exposure scenarios proposed by DOE (DOE/EPA/CDPHE 1996). We have extended the proposed methods in the following ways:

1. Accounting for the large spatial heterogeneity of plutonium and related radionuclide concentrations on and near the Rocky Flats site.
2. Quantifying uncertainty in the environmental models and expressing the radionuclide soil action levels in ways that reflect this uncertainty (e.g., the curves that express the probability of exceeding the dose limit as a function of soil concentration of $^{239+240}Pu$). To the extent possible, the uncertainty distributions are based on site-specific data.
3. Considering exposure scenarios that provide greater exposure opportunity than the ones proposed by DOE, such as the rancher who uses the eastern portion of the site.
4. Including the possible occurrence of a large grass fire sometime within the required 1000-year temporal scope of the assessment. By removing vegetation from a significant fraction of the most contaminated region of the site, such a fire would enhance resuspension of soil-resident radionuclides and make them available for inhalation to people both on- and off-site.

We have implemented these extensions to the extent possible within our time and resources. Our belief is that the information developed in this report makes a strong case for an assessment based on these extensions and gives a good indication of what the results would be. Our principal recommendation is that the work described in this report be strengthened by further research and acquisition of data that could remedy some of the unavoidable limitations. Such research could change the values of the reported RSALs to some degree, and thus we present these numbers as the product of a recommended methodology rather than as final recommendations in their own right. We believe the approach we have developed is sound, and we recommend that it be adopted and built upon in ways discussed below.

12.1 Heterogeneity of Plutonium Concentrations

Heterogeneity of plutonium concentrations in soil on and near the Rocky Flats site is represented by an interpolating model based on fitted power functions in 16 radial sectors centered at the 903 pad. The model was fitted to data from three compilations (Section 4). By its nature, the model is somewhat crude and might be better replaced by a smoothing model based on a kriging scheme or other smoothing interpolation. However, before such a revision is undertaken, we suggest a careful review of all relevant soil data for plutonium, particularly those data for which published documentation is scant or nonexistent. For example, we used some data that were extracted from an undocumented file that M.I. Litaor deposited with the Colorado Department of Public Health and Environment. It might be possible to recover information about these data from RFEETS archives and to adapt additional data from this file for use in characterizing the spatial distribution. Published information of Litaor et al. (1995) makes

extensive use of these data, together with a kriging method, but the paper indicates that no attempt was made to adjust all data to a common basis (sampling depth and time of sampling). We were able to perform the sampling depth adjustment using the work of Webb et al. (1997), but we could not pursue adjustments based on time of sampling. We would like to see a careful analysis of the aggregate of soil data and the fitting of an interpolation model that provides smoothing at the most appropriate scale. We would also be interested in efforts to characterize a generic $^{239+240}Pu$ depth profile over time (1969–mid 1990s). We believe our model is adequate, given the limited amount of data analysis that was possible, but it could be improved.

12.2 Uncertainty

RAC strongly recommends the incorporation of uncertainty analysis into environmental assessments. We consider uncertainty a fundamental part of the modeling process and not an add-on. Uncertainties in model parameters are represented as probability distributions, which are propagated through the model calculations (usually by Monte Carlo methods) to output quantities, such as predicted concentrations in air or food, and to dose and risk.

In performing uncertainty analysis, *RAC* emphasizes the following principles:

A. Uncertainties are represented by distributions of probability. The distributions may apply to single (scalar) numeric variables (the most commonly discussed case) or jointly to multiple variables that may be either stochastically independent or dependent, depending on the interpretation. The distributions can be communicated and explained by various quantitative and graphic devices, such as giving certain percentiles (5th, 50th, 95th) and by showing plotted scatter charts and histograms. Such devices need to be chosen and presented with the background of the audience in mind.

B. *RAC* generally recommends that calculations *not* be deliberately biased high to compensate for lack of knowledge. Rather, analysts should do their best to keep their procedures free of bias. Conservatism, when warranted, should be expressed by increasing the variance of a quantity's uncertainty distribution while keeping its "center" (e.g., 50th percentile) fixed. (The variance is a measure of a distribution's spread or dispersion. The variance is inversely related to the precision with which the quantity is known: if the variance is large, the quantity is known with low precision.) An exception to this general principle occurs in dealing with quantities that are unlikely to affect the outcome of a calculation to a significant degree, in which case the quantities in question may be judiciously biased high.

C. Uncertainties for input variables may be estimated from sample distributions of data, from analytic considerations (e.g., physical arguments that establish bounds for the quantity), by analogy with similar or related quantities, or by seeking consensus of experts. Sometimes nonrigorous arguments based on weight of evidence are persuasive, but when they are offered, they must be acknowledged as such. In doubtful cases, the sensitivity of the outcome to the questioned parameter should be examined; if there is little effect, excessive concern may be unjustified. If there is significant effect, the variance of the uncertainty distribution of the parameter should be increased to a point where there is little doubt that the distribution includes all values applicable. [Note: This statement is strictly applicable only to distributions of random variables with bounded range, such as uniform or log-uniform. In the case of unbounded ranges, the subjective

criterion would have to be put in terms of a high percentile of the distribution.] If such a point cannot be agreed upon, or if the affect on the outcome is so great as to render it virtually meaningless, then further research must be undertaken or alternative simulation strategies must be sought.

D. Results usually should not be presented as point estimates (i.e., single "hard" numbers, such as 2.7 pCi g^{-1}). The desired estimate of the quantity is a distribution, and unambiguous and sufficient information about it should be disclosed (e.g., 5th, 50th, and 95th percentiles; less desirable for nonsymmetric distributions are mean and standard deviation).

E. Explanations should be framed to avoid misunderstandings about the interpretation of statements involving probability.

It is important to clarify the scope of applicability that *RAC* has assumed for uncertainty analysis in this project. When environmental data are considered a sample from population or universe of possible results, the data are treated by conventional statistical methods for making inferences about the source population. The results may be used to postulate distributions for parameters in models of environmental processes. In this sense, the environmental data and the natural processes that affect them are treated as uncertain. On the other hand, we do not usually treat scenario parameters (e.g., the breathing rate or dietary habits of a subject) as being uncertain. This dichotomy has been challenged by reviewers of draft project reports. We offer here a more complete explanation than we have given in the past, in the hope that readers will be able to consider our reasons in their entirety.

The environmental models and parameters represent something that we do not control. For the most part, this "something" is the natural environment (or a very restricted part of it), but it can also include anthropogenic processes such as a source term. The models represent this environment as a system of state variables, including those that stand for concentrations of radionuclides in soil, air, and so on. We attempt to estimate the past or predict the future of this system, and to quantify uncertainties about those estimates or predictions (generally we say "predict" in either case). The representations of uncertainty are themselves models, and their application includes subjectivity.

The scenarios for radiation protection, on the other hand, are under our explicit control. They are hypotheses that we set; their subjects are not real people. They provide a means of constructing criteria for interpreting the predicted (or measured) radionuclide levels in environmental media. Prospective calculations that we perform are really about the environmental media. But they are expressed in terms of dose or risk to a scenario subject to place them on a more meaningful (and lower-dimensional) scale.

Probability distributions associated with the environment, which we do not control but which we must somehow simulate, are of a different character from distributions associated with variability within populations from which a scenario subject is imagined to have been drawn. It seems to us generally confusing, and possibly misleading, to mix the two kinds of probabilities together in order to make uncertainty statements about exceeding dose limits. It seems to us much clearer to choose our scenario subjects with fixed numeric properties (breathing rates, dietary habits, and behavioral characteristics) that would be protective of a reasonable fraction of the population from which we assume the subjects come. If multiple properties are involved, then we obviously cannot set them all at the 95th percentile and assert that we are conservative for only

95% of the population. But we do believe that they should be set to fixed values, in such a way as to define the subject as being credibly protective of an acceptable proportion of the population. Certainly, it is always reasonable to review an assessment as a whole and ask whether too much conservatism might have been introduced. But care has to be taken in doing so.

Note that when a millennium is the time domain of a prospective study (as is the case for this project), the scenario becomes a succession of hypothetical individuals, all having similar location and characteristics, but with their exposure environment evolving from generation to generation. If one were to treat the scenario subjects statistically, would successive generations be stochastically independent with respect to their physical and behavioral properties? Or would one consider autocorrelations, to account for family traits in different generations? Or would we sample one set of properties at each Monte Carlo realization and apply them equally to all generations? Many questions of this kind can be raised to illustrate the conceptual problems that arise when one ventures down the path of "realism" expected from treating scenario subjects as samples from real populations.

Finally, it might be helpful to the reader to contrast the situation described above, for prospective assessments, to retrospective studies such as dose reconstructions. If a risk analysis is carried out for such a study, the affected populations are real, and distributions of properties of those populations can (at least in principle) be estimated (e.g., by Census statistics and sampling surveys). We can then quite reasonably consider these distributions as part of the total uncertainty in the risk estimate and combine them with distributions of concentrations in exposure media. The outcome, for example, might be the number of health effects that would be predicted to result from the collective exposure under study. This number is uncertain, not only because of our uncertain predictions of environmental concentrations, but also because of variability within the affected population with respect to our determination of the relevant properties (e.g., breathing rates, diet, proximity to contaminated media). In a retrospective study, we do not have the luxury of defining a hypothetical individual whose properties suffice to protect most people who *might be* exposed. The purpose of a dose reconstruction is not to protect anyone, but rather to study potential or realized effects of what has already happened.

RAC recommends that uncertainty analysis be retained in applications of the methodology described in this report, and that it conform to the spirit of principles A–E above.

12.3 Scenarios

RAC has made calculations for scenarios other than those discussed in DOE/EPA/CDPHE 1997. It is not our purpose to recommend that particular scenarios be adopted for calculating the RSALs that govern the cleanup of the Rocky Flats site. The question of final determination of scenarios is closely related to discussions and decisions about the subsequent use of the land and the durability of any institutional controls that might be proposed. Such matters have a political component that is beyond our scope.

We have recommended by our example, however, that additional scenarios be explored and considered in the discussions by the community and the decision-makers. The details of these scenarios are given in our Task 3 report (Aanenson et al. 1999). Generally, the scenarios explore broader opportunities for exposure than the ones proposed by the agencies. The *RAC* Rancher scenarios are very much in the tradition of regulatory radiological assessment practice. It is partly for this reason that we have emphasized them in this report. But we encourage all interested

parties to consider other exposure scenarios and variants that are deemed relevant to anticipated long-term use of the land.

12.4 Future Grass Fire

The question of the possible effects of a future wild fire has inspired considerable discussion during the completion of Task 5. In the stochastic fire model described in this report, the magnitude of the effect depends critically on a highly uncertain parameter, namely the resuspension flux for the large devegetated area following a fire. Generically, we have taken the median ratio of the flux from a completely devegetated area of 10^7 m^2 divided by the corresponding flux from the same normally vegetated region to be 200 (Section 5). We have reduced this ratio by multiplying it by stochastically modeled mitigating factors based on size of the burn area, distance of the fire from the subjects, and time of regrowth. The baseline value of 200 could suggest too much importance for a future fire, but it could also understate the effect. We recommend additional research to try to find useful data from applicable studies that could reduce the uncertainty in this parameter. It may be that some new field measurements above recently burned areas in the Front Range region could be supported, but relatively large areas would likely be more useful than small experimental tracts. If such measurements are undertaken, they should include monitoring of meteorological variables together with fluxes and air concentrations with particle size distributions.

The effect of future periods of drought on the probability of a fire in a specified year is a question we have not been able to consider explicitly, but this probability could be an important parameter in the stochastic fire model. Periods of drought or greater rainfall could accompany future climate change. Vegetation could take more than one year to recover if a fire occurred during an extended drought, and the frequency and size of fires would be greater. A recovery time larger than one year would not affect an RSAL calculation based on a maximum annual dose criterion, but it could be significant for a limiting criterion based on lifetime risk resulting from the total lifetime exposure of a scenario subject. Some additional computer studies could give a better sense of the sensitivity of RSALs to the assumptions under the different limiting criteria.

An aspect of the fire that we have had to neglect is the effect of contaminated smoke particles on people who might have breathed them at the time of burning. This is an important unanswered question, for which the most important ingredients of an answer are available (e.g., Little 1980). However, to attempt it would have required a more elaborate scenario structure, and it is not clear just how the analysis would have fitted into the RSAL scheme. Some modifications of our dispersion model would also have been required. These difficulties can undoubtedly be worked out, but we regrettably have to leave this important question for now as a recommendation.

12.5 Other Recommendations

12.5.1 Groundwater pathway

Inclusion of the groundwater pathway has a small effect on the simulation results for plutonium isotopes. RAC's analysis for Task 3 (Aanenson et al. 1999) indicated that the soil-water equilibrium distribution coefficients (K_d) for plutonium and americium were large. These

large magnitudes indicate that transport of these elements from the surface soil through to the groundwater aquifer is slow, with very little material reaching the aquifer within the 1000-year temporal scope of the assessment. Thus, very little of the plutonium and americium from the surface soil reaches the scenario subjects by way of the groundwater pathway.

However, we stress that the groundwater pathway is complex, whereas its treatment within RESRAD is simplistic and may not adequately represent the contamination of groundwater over time. Specifically, colloidal transport and oxidation-reduction reactions under anaerobic conditions have been postulated as processes that can significantly enhance the mobility of plutonium (see Appendix B, "Alternative Groundwater Dose Calculations"). Future research into vadose zone transport processes at Rocky Flats or new information about plutonium geochemistry and site-specific distribution coefficients could affect these results and should be taken into account in any future dose assessment. In the event that contaminants are found to move more rapidly through the vadose zone into groundwater, we believe the effect on the calculated plutonium isotope RSALs would still be small. Rapid transport by way of groundwater pathways would imply faster depletion of surface contamination, and the increase in groundwater concentration would likely be offset by the diminished surface soil concentration. Therefore, although changes in estimated groundwater transport of radionuclides may occur as better information is developed, we believe these changes will likely be insufficient to cause the dose limit to be exceeded.

In the case of uranium isotopes, RSALs tend to be more sensitive to the groundwater pathway, and in fact, they control the RSAL at the 10% probability level. Again, we recommend additional research into the mobility of uranium at the RFETS, coupled with environmental monitoring of groundwater for uranium isotopes.

12.5.2 Probability level for exceeding the dose limit

We have presented results for each scenario and annual dose limit as a plot of the probability of exceeding the dose limit, expressed as a function of $^{239+240}$Pu (or ^{238}U) soil concentration. The radionuclide soil action level can be obtained from such a plot as the concentration that corresponds to a specified probability level. We have shown RSALs corresponding to a 10% probability of exceeding the annual dose limit, and this value is consistent with EPA guidance for CERCLA cleanup activities at Superfund sites. The following quotations (which were pointed out by a reviewer of the draft Task 5 report) indicate the nature of this guidance:

> "...actions at Superfund sites should be based on an estimate of the reasonable maximum exposure (RME) expected to occur under both current and future land use conditions. The reasonable maximum exposure is defined here as the highest exposure that is reasonably expected to occur at the site..." (*Risk Assessment Guidance for Superfund, Volume 1, Human Health Evaluation Manual (Part A)* Interim Final, EPA-502/1-88-020.)

> "The high-end of the risk distribution is, conceptually, above the 90th percentile of the actual (either measured of estimated) distribution. The conceptual range is not meant to precisely define the limits of this descriptor, but should be used carefully by the assessor as a target range for characterizing 'high-end' risk." ("Guidance on Risk

Characterization for Risk Managers and Risk Assessors," Memo from Henry Habicht II, Deputy Administrator, EPA, to Assistant Administrators and Regional Administrators, Febuary 26, 1992)."

The use of 5% or 10% as a default measure of relative smallness is common in scientific practice. As one may see from nearly any statistics textbook, 90% confidence intervals are typically derived, and tests of hypothesis usually seek to reject the null hypothesis at the 5% level. There are, of course, circumstances that indicate more (or sometimes less) stringent criteria.

RAC recommends that a 10% criterion be applied to estimating RSALs from the probability curves given for each scenario and dose limit.

12.5.3 Applicable dose limits

Annual dose limits of 15 mrem year^{-1} (unrestricted use) and 85 mrem year^{-1} (restricted use) were specified in the contract, and *RAC* is required to provide RSAL estimates for both levels. However, we believe that the inclusion of 85 mrem year^{-1} standard in future discussions should be reconsidered by DOE and the oversight panel. This dose limit appears to be inappropriate for future uses of this site within current regulatory guidance. A reviewer of the draft reports for Tasks 2 and 5 of this project provided the following background, which is part of the document review record, and we quote it here because of its importance:

> "The 85 mrem/y dose criterion was proposed by EPA as a supplementary upper bound on the possible exposure of individuals in order to assure a minimum level of protection in the event of *unanticipated* failure of institutional controls, not as an alternative dose limit. Further, such failure was expected normally to be of short duration, because it was assumed to be corrected when identified. The criterion was not intended for application to planned long-term uses when institutional controls are assumed (i.e. planned) to no longer exist (as in the three DOE scenarios noted above) and *it was certainly never intended for use as a occupational standard*, as it is used in the *RAC*-4 scenario. The Superfund does not recognize different risk (or dose) criteria for individuals exposed as workers vs. other members of the public after a site has been cleaned up. The only way an increased dose to a worker over that permitted any member of the public would be permissible is for the situation in which the worker is exposed to be the result of licensees activities involving radiation as a part of the work product. Of course, at an industrial site, it is appropriate to take account of the decreased residency of a worker, as was done in scenario *RAC*-4. However, *the dose criterion that should be applied in RAC-4 is 15 mrem/y, not 85 mrem/y*. We note that, in the current directive under which EPA regulates radiation cleanups (OSWER Directive No. 9200.4-18; August 1997), the 85 mrem/y criterion has been dropped entirely, since it is assumed to be unnecessary. . . .
>
> ". . . the viability of any industrial scenario depends on the guaranteed continued effectiveness of institutional control. It remains not obvious to this reviewer that either the commitments or assurance of effectiveness for the necessary institutional control exist. The DOE report [DOE/EPA/CDPHE 1997] depends on the "Rocky Flats Vision"

for assurance of such control. This document was not available for review. However, a "vision" is not a legal commitment, and the discussion of near and immediate term land uses and, more significantly, the absence of any discussion of long-term land use (e.g. in the last paragraph on p. 6-15 of the DOE report) creates the impression that the state of commitments for and assurance of effectiveness of institutional controls in the future is very uncertain. If the lead agency (DOE), State, and local officials cannot provide reasonable assurance of maintaining effective institutional control for 1000 years, then consideration would have to be given to cleanup of the site to 15 mrem/y under scenarios that do not depend on the presence of such control. . . ."

RAC recommends that the relevance of the 85 mrem year^{-1} standard be reconsidered in the light of contemporary regulatory guidance and contemplated uses of the Rocky Flats site.

12.5.4 Alternate RSAL calculation method

Section 7 describes an alternative calculation method for plutonium RSALs for the Rocky Flats site. This method is labor intensive and difficult to automate; otherwise, we would likely have developed it for primary use in the RSAL calculations for this report. It has the advantage of a simple interpretation that ties the RSAL directly to the modeled remediation strategy, and it takes into account $^{239+240}$Pu soil concentration as a function of location. The extended sum-of-ratios method used for most of the report depends principally on the plutonium air-to-soil concentration ratio, a fact that can create confusion for the interpretation of the RSALs. We have applied the alternate method to the *RAC* Rancher scenario and have found that it produces comparable RSALs for that scenario when the receptor is placed on the site. However, when the receptor for the same scenario is located just east of Indiana Street, the result of the alternative calculation indicates that no remediation would be required to meet the 15 mrem year^{-1} dose limit. The alternate method is more explicit than the extended sum-of-ratios method, and its interpretation is straightforward. *RAC* recommends that this method be developed further for supplementary (or possibly principal) use in any further scenario analyses or future dose assessment.

12.5.5 Lifetime risk criterion for RSALs

Some reviewers of the draft Task 5 report questioned the use of a maximum annual dose criterion for limiting radionuclide soil concentrations, suggesting that to proceed directly from a lifetime risk criterion would be preferable. In calculating RSALs, *RAC* was constrained by contractual agreements to apply 15 mrem maximum annual dose limits for scenarios involving full public access to the site. (Elsewhere in this section, we recommend that elimination of the 85 mrem annual limit be considered.) But we are open to the view that such a risk-based approach might be appropriate.

The recent Federal Guidance Report 13 (Eckerman et al. 1999) provides lifetime risk-per-unit-exposure factors for the relevant pathways that would facilitate an approach based on direct lifetime risk limitation, which as a technical matter could be carried out with some modification of the scripts we have used for the calculations reported here. However, there are serious questions about the role of uncertainty in the results when uncertainty for risk coefficients is

greater than that for dose coefficients. In the present work, dose coefficients (dose conversion factors) have been treated as scenario parameters, and accordingly as fixed quantities. The risk coefficients could be treated similarly, leaving the results conditional given the values of the risk coefficients and subject to interpretation in the light of what is known about the uncertainties in these parameters. It is also possible that the uncertainties in dose and risk coefficients could point to more conservative RSALs than the ones we have estimated. *RAC* recommends that this question receive further study as the cleanup of the Rocky Flats site is discussed.

13. CONCLUSIONS

The primary objective of this project has been to review radionuclide soil action levels (RSALs) adopted by the Department of Energy, the Environmental Protection Agency, and the Colorado Department of Health and Environment in 1996 for cleanup at the Rocky Flats Environmental Technology Site (DOE/EPA/CDPHE 1996). Another objective has been to recommend a technical approach for independently deriving RSALs for the site. We applied this approach to the Rocky Flats data using the most restrictive exposure scenarios approved by the Oversight Panel and assuming a 10% probability that the 15 mrem per year dose limit will be exceeded (i.e. a 90% probability that the dose limit will not be exceeded). Using this approach, the technically derived RSAL for $^{239+240}Pu$ in soil at Rocky Flats would be 35 pCi g^{-1}. This calculation was corroborated by an alternate method calculation that also resulted in an RSAL at the 10% level of about 37 pCi g^{-1}, suggesting 35 pCi g^{-1} as a technically based RSAL for the Rocky Flats site. The results as presented are a reasonable indication of RSAL magnitudes based on purely scientific considerations if the prescribed dose is not to be exceeded.

The calculation of uranium RSALs was done somewhat differently than those for plutonium because of significant differences in the nature and extent of contamination and the mobility of uranium in the subsurface. For each uranium scenario, consideration was given to whether groundwater was a viable pathway. A viable groundwater pathway assumed that the surficial aquifer underlying the site would provide enough water for human consumption and irrigation. The impacts of a probabilistic fire were also evaluated but inclusion of this process in our calculations made little difference in the resulting RSALs. Assuming the groundwater pathway was viable and a 10% probability, the technically derived ^{238}U RSAL for the most restrictive scenario (the rancher child) was 10 pCi g^{-1}.

We believe the general approaches presented in this report and these results are sound and we recommend their adoption. Data limitations impose uncertainties on estimates of doses, and we have been careful to indicate these uncertainties in our analysis. The project's time and budget goals precluded a more in-depth investigation of several important areas of research that, if addressed in the future, could strengthen this analysis. We have presented these recommendations for further research and recognize that they could change these results somewhat and improve them as a basis for decision making.

Our methodology is based on several extensions of an earlier approach proposed by DOE/EPA/CDPHE (1996) that used the RESRAD computer program. The contract required that the work consider maximum annual dose limits of 15 and 85 mrem in any year over the next 1000 years. We adopted the 15 mrem per year limit for a technically based RSAL because it is more protective of the public and because our evaluation of risk associated with this dose better corresponds to the target level of risk associated with federal guidance (e.g. CERCLA). Although we considered several computer codes to use as the basis of our analysis, the RESRAD code was adopted because it was the most practical choice and because we were required to make calculations with RESRAD in addition to any other code that may have been selected. Therefore, we designed extensions to RESRAD to include (1) consideration of the heterogeneity of radionuclide concentrations in soil around the site, (2) quantifying uncertainty in predictions of dose, (3) consideration of additional exposure scenarios, and (4) treating the possible occurrence of a large grass fire.

Other factors beyond the scope of this work should be considered in the selection of cleanup strategies for Rocky Flats. The soil action level that is applied for cleanup should be decided by federal and state authorities and the community working together to arrive at a cleanup level that provides long term protection of the public. Figure 13-1 shows probability curves for the most restrictive scenarios. This figure broadly summarizes the results of our work. Parties involved in the decision process might find the figure useful in their deliberations keeping in mind the different exposure scenarios represented by the curves and the uncertainties involved.

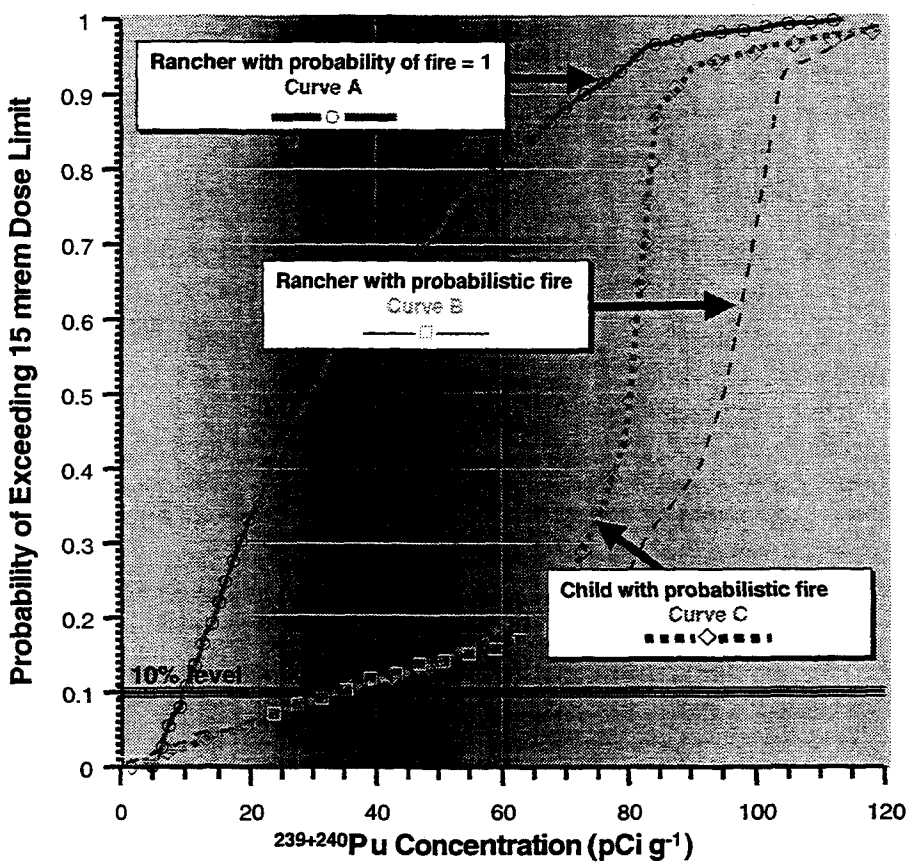

Figure 13-1. Composite graphic illustrating the most restrictive scenarios and showing a region centered at a soil action level of 35 pCi g^{-1}. Curve A represents the rancher and assumes that a fire occurs with a probability of 1; curve B represents the rancher scenario and takes into account the occurrence of a fire as a probabilistic event; curve C represents the child scenario and, like curve B, incorporates the probability of a fire.

There are several features illustrated in this figure that are important to note. Curve A, defined by the rancher scenario and with the probability of a fire equal to one, likely represents the most conservative set of assumptions and hence the most restrictive radionuclide soil action level. We say "likely" because further research into the impacts of a prairie fire could show that we have underestimated the effects of the fire. Curve B represents the rancher and incorporates a stochastic model of a future fire. With our assumption of a 10% probability of exceeding the dose

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limit, this curve yields a soil action level of about 35 pCi g^{-1} (the exact value is 33 pCi g^{-1}). Toward the left of the curve, the shape and slope are controlled primarily by inhalation and the probability of occurrence and extent of a fire. However, as the soil concentration of $^{239+240}Pu$ increases, the contribution to dose from ingestion becomes more prominent, and the slope is more influenced by this pathway. Curve C is that of the rancher's child with the stochastic fire model included. This curve is quite similar to that of the rancher with the stochastic fire model but the curve indicates this scenario is not as protective as the rancher scenarios in the region of lower RSAL concentrations. At higher RSAL concentrations, however, this curve becomes more protective than that of the rancher because the ingestion pathway becomes more influential. The steepness of the curve reflects less uncertainty in the calculation. The rancher scenario with the probabilistic fire is our basis for selecting an RSAL at the 10% probability level.

To give a better visualization of our results, we have underlain Figure 13-1 with a spectrum that expands in both directions around 35 pCi g^{-1} which is about where the rancher and child of the rancher curves intersect the 10% probability level. Colors are darker near the center of the spectrum and lighter farther out. It is important to understand that curves A, B, and C are based on a sum-of-ratios calculation that incorporates the contribution to dose from other radionuclides present in the soil in addition to $^{239+240}Pu$. The graphic suggests a technically based RSAL of about 35 pCi g^{-1} at the 10% probability level and a range of possible RSALs in both directions centered at this value. Although there is no quantitative basis for the boundaries of this range, it is apparent that going too far in either direction from the center of the spectrum can potentially be problematic for a variety of reasons. Radionuclide soil action levels that are significantly lower may correspond to unrealistically conservative scenario descriptions, which could lead to significantly greater cleanup costs than can be justified. On the other hand, RSALs that are significantly larger lead to a high probability of exceeding the prescribed dose limit and could impact human health. It is especially important to understand that the calculation based on the child scenario and influenced primarily by soil ingestion is scientifically well supported. It is unlikely to change greatly unless values for important parameters change, such as the dose conversion factors or the soil ingestion rate. Therefore, curve C effectively represents an upper bound for the RSAL. If the soil action level were too close to this curve, the probability of exceeding the dose limit is greatly increased.

We also developed an alternate method for calculating acceptable levels of radionuclides in soil. This method was based on calculating annual doses to the receptor for different remediation (i.e., cleanup) levels. The remediation level that resulted in a 10% probability that the 15 mrem dose limit would be exceeded defined the RSAL. This method more explicitly addresses the heterogeneity of the site and makes it possible to estimate RSALs that correspond more directly to a remediation strategy than does the sum-of-ratios technique used with RESRAD. The approach is more difficult to implement and therefore has not been fully automated in the analysis. However, because it is more explicit, it is a useful check on the sum-of-ratios method, and we include its results in these conclusions. This alternate calculation resulted in an RSAL at the 10% level of about 37 pCi g^{-1}, suggesting the value of 35 pCi g^{-1} should be strongly considered as a technically based RSAL for the Rocky Flats site.

Our analysis is based on the best available data and methods that we could employ. During the course of our work, we have identified important research that should be completed in order to strengthen our methodology. In addition, changes in the design specifications or scenario assumptions on which this methodology is based would change the results accordingly. This

flexibility is quite important to keep in mind because a number of issues that could affect these results have been raised during the course of our work.

While our methodology and the resulting RSAL values are scientifically defensible and are based on sound science, *RAC* believes that additional work could reduce some of the uncertainties and refine the RSALs. There were specific areas where more information or more organized research and scientific inquiry would have allowed us to make better estimates of parameters or to develop more well-defined methods in our approach. Foremost among these are data that quantify the impact of a prairie fire on the land now occupied by the Rocky Flats site and the data from the Actinide Migration Evaluation studies. Other important areas include:

- effect of prairie fires on the resuspension of material
- time sequence of revegetation following a natural event like a fire
- more realism in the resuspension model for RESRAD
- developing a methodology to estimate the effects of combined exposure to both the uranium hotspots and the widespread plutonium contamination at Rocky Flats
- construction of a computer-implemented model of the Rocky Flats to permit flexibility in analyzing different radionuclides, sources, and pathways
- groundwater transport properties at Rocky Flats
- new discoveries about site-specific distribution coefficients
- potential for accumulation of actinides on offsite lands and water resources
- protection from violation of the Rocky Flats Cleanup Agreement (RFCA) surface water standards for plutonium.

A sound technical foundation and credible scientific methodology are the most important elements in setting soil action levels for Rocky Flats site. However, the final decision on setting the RSALs ultimately lies in the hands of the stakeholders, DOE, and other State and federal authorities. There are other criteria that influence the decision-making process for the Rocky Flats site, such as the cost of cleanup, protection of ecological resources, and community values. The approach to cleanup that is ultimately implemented by the DOE at the RFETS will involve many political, social, economic, and moral decisions. It is imperative that all involved in the decision process recognize these factors and the integration of ideas that must go into making a decision of this type.

RAC's task was to evaluate the RSALs adopted for Rocky Flats in 1996, to develop a methodology for independently determining RSALs, and to calculating RSALs for Rocky Flats by applying this methodology. We conclude that applying our method to the exposure scenarios approved by the Oversight Panel, using 15 mrem as a dose limit, and assuming a probability level of 10%, indicates a technically based RSAL for $^{239+240}Pu$ in soil at Rocky Flats of 35 pCi g^{-1}. For uranium, a technically derived RSAL using our methodology and assumptions would be 10 pCi g^{-1}.

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APPENDIX A

COMPUTATIONAL DETAILS

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COMPUTATIONAL DETAILS

This appendix illustrates how RESRAD was used in conjunction with PERL script files to perform Monte Carlo simulation for the radionuclide soil action levels. In the calculations we performed, we bypassed the graphical user interface provided in the RESRAD distribution files and instead wrote RESRAD input files and executed the main computational unit of RESRAD (RESMAIN3) from the PERL script. The primary functions of the PERL script include:

- assign values for all RESRAD input parameters
- sample those parameters that were treated stochastically
- write RESRAD input files
- execute RESMAIN3
- extract doses and single radionuclide guidelines for each nuclide from the SUMMARY.REP file and save to separate files.

Each scenario was run using a separate PERL script file and uranium and plutonium isotopes were run separately. As discussed in Section 4.2, uranium was treated differently because the nature and extent of uranium contamination was different from that of plutonium contamination.

The default dose conversion factor library from RESRAD is based in the International Commission on Radiation Protection (ICRP) publication 30. Therefore, new dose conversion factor libraries were written to accommodate age-dependent dose conversion factors from ICRP 70. Three separate dose conversion factor files were written; one for the infant scenario (INFANT.BIN), one for the child scenario (CHILD.BIN), and one for the remaining scenarios that involved adults (ADULT70.BIN). The dose conversion factor file used in the calculation is identified in the PERL script.

An external C program (FIRECONC.EXE was used to calculate the fire probabilities and is called in the Perl script before the start of the Monte Carlo simulation. Several other PERL scripts were used to take the single radionuclide soil guidelines and compute the probability distribution of the sum of ratios. The DO_GRAPH.PL script opens the file containing the single radionuclide soil guidelines and calls the scripts, SRATIO.PL and CUMPROB.PL to compute the sum or ratios and the probability that the sum of ratios exceeds the specified dose limit.

Illustrated below is the PERL script file for resident rancher with the water pathway off followed by the DO_GRAPH.PL, CUMPROB.PL, and SRATIO.PL. Comments are provided throughout the script to aid the reader's understanding of the process.

```
# racrnc:pl (RANCHER Adult), scenario beginning 2000 with water pathways off -
no fire.
# PERL program for executing RESRAD v. 5.82 in Monte Carlo mode
# with the Rocky Flats radionuclides.
#
# Please note that this program works only with a very restricted setup.
# The first statement (require) must point to a directory that contains
# the PERL file sample.pl. The variables $RESRAD_path and $Working_directory
# must contain correct path representations to the RESRAD 5.82 directory
# and the user's working directory, respectively. The working file
# indicated by the variable $radfile must reside in the user's working
```

```
# directory (the program will make numerous substitutions in it). The files
# RFSAL.B1, RFSAL.BIN, RFSAL.D1, RFSAL.D34, RFSAL.D5, RFSAL.RN, and
# RFSAL.SF packed with this script must be placed in the RESRAD 5.82
# directory.
#
# Output files are $outdose = mcdose.prt and $outsal = mcsal.prt (these
# settings can be changed; see below). These files are written in the
# working directory. They have headers that may need to be removed for
# processing with other software for plotting histograms or computing
# percentiles or other statistics.

# List of principal radionuclides
@nuc_name = ("Am-241","Np-237","Pu-238","Pu-239","Pu-240","Pu-241","Pu-242");

require "c:\\rfpsal\\binfiles\\sample.pl";   # distributions for Monte Carlo
sampling

srand(314159265);            # set randon number seed

$RESRAD_path = "c:\\RESRAD";
$Working_directory = "c:\\rfpsal\\task5\\rev2\\RACRNC";

#-------------------------------------------------------------------
# Parameters that will most often be changed for a given scenario
#-------------------------------------------------------------------

# Year-2000 soil concentration distribution for Pu-239+240 (pCi g^(-1))

$Pu_in_soil_GM = 1.81;
$Pu_in_soil_GSD = 4;

# Year-2000 air concentration distribution for Pu-293+240 (pCi m^(-3))

$Air_conc_Pu_GM = 2.33E-5;
$Air_conc_Pu_GSD = 4;
#***** Parameters for resuspension flux ratio calculation
$Air_conc_Pu_fire_GSD = 16;       # when fire is indicated, this GSD
              # will be used for the uncertainty of the air concentration
              # of Pu (this is attributed to the estimated soil flux)
# Burn-areas for wildfires in Arapaho-Roosevelt National Forest and Pawnee
# National Grasslands, 1900-1998 (acres) (data sorted into ascending order)
@Burn_area = (100, 124, 130, 140, 141, 150, 167, 170, 177, 180, 182,
    200, 210, 220, 230, 235, 250, 256, 256, 275, 283, 300, 312, 364,
    370, 370, 390, 400, 450, 470, 477, 477, 500, 500, 606, 660, 715,
    740, 748, 757, 1000, 1007, 1008, 1148, 1190, 1200, 1214, 1221,
    1804, 1967, 2471, 2635, 2734, 2800);
# Total area of Arapaho-Roosevelt National Forest and Pawnee National
# Grasslands (acres)
$Park_area = 1.5E6;

$pmin = 0.0012; $pmax = 0.0023;
              # about 1 fire per 1000 years at the RF site; based on fire
              # statistics in the Arapaho-Roosevelt National Forest
              # (1.3 million acres) and the Pawnee National Grasslands
              # (193,000 acres), using fires that burned 100 acres
              # or more between 1900 and 1998.

$flux_ratio_0 = 200;    # baseline ratio of median soil flux with and
                     # without fire
$year_of_fire = 1;  # parameter to be reset by fire_flux()
              # subroutine, to give the
              # year of the fire that produces the maximum
```

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```
                # Pu flux estimate
#***** End of parameters for resuspension flux ratio calculation

#-----------------------------------------------------------------
# Working files
#-----------------------------------------------------------------
$radfile = "mcres.rad";           # NAMELIST INDATA master file
$xxfile = "mcresxx.rad";   # copy of modified $radfile to \resrad
$outdose = "c:\\rfpsal\\task5\\rev2\\racrnc\\mcdose.prt";       # output file
for Monte Carlo dose results
$outsal = "c:\\rfpsal\\task5\\rev2\\racrnc\\mcsal.dat";         # output file
for Monte Carlo SAL results
$dffile = "ADULT70.BIN";          # binary library for dose conversion factors
$dmpfile = "c:\\rfpsal\\task5\\rev2\\racrnc\\output.dmp";          # dump file
of sampled values
open (DUMP, ">$dmpfile");
#-----------------------------------------------------------------
# Set the number of Monte Carlo realizations ...
# if $nmc = 1, the median or mean will be used for any sampled
# parameter (median for logarithmic distributions), mean for others.
#-----------------------------------------------------------------

$nmc = 500;     # number of Monte Carlo iterations (=1 for deterministic)

$Title = "Resident Rancher (Adult), composite fire scenario with water pathways
off";

#----- More fire parameters
#
# Receptor coordinates (km UTM) Located at the industrial worker site

$x_R = 486.1;
$y_R = 4415.310;

# Rectangular fire domain (roughly rancher's land) (km UTM).
$xmin = 483.2;
$ymin = 4413.12;
$xmax = 485.96;
$ymax = 4417.23;

# Open the stream of normalized air concentrations from future fires.
# The worst case corresponds to 1.0, and leaching of Pu from the soil
# is not yet taken into account. In subroutine %fire_flux, each sampled
# value is multiplied by $flux_ratio_0, making this the maximum value.
# Then a random year is generated, and the product is adjusted for
# leaching. If the result is > 1, it is multiplied by the default
# flux (i.e., the one for standard ground cover), and the realization
# corresponds to a fire. Otherwise, the default flux is used and the
# realization coresponds to the year-2000 scenario.

$nfs = 1000; # sample size from FIRESTREAM
open(FIRESTREAM,
    "c:\\rfpsal\\binfiles\\fireconc.exe $nfs $x_R $y_R $xmin $ymin $xmax $ymax
|")
    || die "Cannot execute fireconc";
print STDERR "Generating a sample of $nfs observations from fireconc --\n";
print STDERR "this may take a minute or so ... ";
for (1 .. $nfs)
{
    $line = <FIRESTREAM>;
    chomp $line;
    $line =~ s/[ ]+//g;
```

```
    push @firestream, $line;
}
print STDERR "sample complete and stored.\n";
close FIRESTREAM;
# End new fire stuff
#------

# Set $Version to "old" or "new" to indicate which area factor
# will be used (v. 5.61 or 5.82, respectively).
$Version = "new";

# Annual dose limit
$Dose_limit = 15;   # mrem/year

# Areas (m^2)
$Area = 1E7; # Area of contaminated zone
$Watershed_area = 8.28E6;  # Watershed area for nearby stream or pond

# The area of the fire domain generally should not be changed.
# The value shown represents most of the eastern end of the site from
# just west of the 903 pad to Indiana Avenue.
$Fire_domain = 1E7;

#-------------------------------------------------------------------
# Pathways to be considered: set each indicator variable equal to 1
# to include a pathway and to 0 to exclude it.
#-------------------------------------------------------------------

$i_ground = 1;
$i_inhalation = 1;
$i_plant = 1;
$i_milk = 1;
$i_meat = 1;
$i_aquatic_food = 0;
$i_drinking_water = 0;
$i_soil_ingestion = 1;
$i_radon = 0;

$select_path = 1 * $i_ground + 2 * $i_inhalation + 4 * $i_plant
    + 8 * $i_milk + 16 * $i_meat + 32 * $i_aquatic_food
    + 64 * $i_drinking_water + 128 * $i_soil_ingestion
    + 256 * $i_radon;

# The simulation will begin in 1971, the year for which we have the
# proportions of Pu and Am isotopes from Krey et al.

#-------------------------------------------------------------------
# Soil data
#-------------------------------------------------------------------

# The following isotopic activities (pCi) are based on a unit mass of
# total Pu in 1971 (100 g). Am-241 was calculated from the decay chain,
# beginning with 1 unit of Pu-241 in 1958 and no Am-241 and integrating
# the Bateman-type equation to 1971, then using the computed Am-241:Pu-241
# ratio and the given Pu-241 activity in 1971 to estimate Am-241 activity
# in 1971. These are normalized initial activities. The "true" initial
# activities will be based on these values and the "true" Pu-239+240
# soil concentration for the 1990s, from which the 1971 value is back
# calculated, allowing for leaching. From this value, the remaining
# initial values are computed.

%ActNorm0 = ( "Pu238", 1.164E11, "Pu239", 5.901E12, "Pu240", 1.103E12,
```

```
    "Pu241", 2.255E13, "Pu242", 5.330E7, "Am241", 5.458E11 );

# The actual initial values (pCi g^(-1)) will be calculated and
# put into the following associative array by the program:
%Act0 = ( "Pu238", 0, "Pu239", 0, "Pu240", 0, "Pu241", 0, "Pu242", 0,
       "Am241", 0);.

# Specific activities (pCi g^(-1)) given for reference:
#%SA = ( "Pu238", 1.714E13, "Pu239", 6.217E10, "Pu240", 2.279E11,
#    "Pu241", 1.030E14, "Pu242", 3.919E9, "Am241", 3.433E12 );


# Initial values:
#
# Pu-239+240 in soil distribution for the year 2000
#$Pu_239_240_GM = 17.1;    # geometric mean (pCi g^(-1))
#$Pu_239_240_GSD = 1.26;   # geometric standard deviation
$Pu_239_240_GM = $Pu_in_soil_GM;
$Pu_239_240_GSD = $Pu_in_soil_GSD;


# Uncertainty factor for 1971 Am-241:Pu-239+240 ratio
$Am_Pu_239_240_GM = 1;
$Am_Pu_239_240_GSD = 1.27; # Using .13+/-.03 from Krey et al.,
       # we estimate GSD = 1 + 13/3 = 1.27

# Uncertainty factor distributions for Kd (cm^3/g)
# partition coefficients for Pu, Am, and U
# (treated as independent):
$Kd_Pu_GM = 2300;
$Kd_Pu_GSD = 5.6;
$Kd_Am_GM = 1800;
$Kd_Am_GSD = 8.1;
$Kd_U_GM = 2.3;
$Kd_U_GSD = 5.4;

# Uncertainty distribution (uniform) for total porosity:
$eps_tot_min = 0.3;
$eps_tot_max = 0.3;

# Uncertainty distribution (uniform) for effective porosity:
$eps_eff_min = 0.1;
$eps_eff_max = 0.1;

# Hydraulic conductivity of the contaminated zone (m/y):
$Hydraulic_conductivity = 44.5;

# Contaminated zone b parameter:
$b_parameter = 10.4;

# Uncertainty distribution (uniform) for precipitation rate (m/y):
$Precip_min = 0.381;
$Precip_max = 0.381;

# Runoff coefficient
$Runoff = 0.2;

# Transpiration coefficient
$Evap_transp = 0.92;

#-----------------------------------------------------------------------
# Food and water parameters
```

```
#-----------------------------------------------------------------------

# Irrigation rate
$Irrig = 0;     # m y^(-1)

# Soil bulk density
$rho_b = 1.8; # g cm^(-3)

$Delta_z = 0.15;    # thickness of contaminated zone (m)
$Mixing_depth = 0.03;      # depth of soil available for resuspension (m)
$Erode_CZ = 0;             # erosion rate of contaminated zone (y^(-1))
$Root_zone = 0.9;   # depth of root zone (m)

$Dilution_length = 3;      # mixing model parameter (m); 3 is RESRAD default
                    # for the old area factor computation

# Dietary intakes
# DIET(1) -- Fruit, nonleafy vegetables, grain (kg y^(-1))
# DIET(2) -- Leafy vegetables (kg y^(-1))
# DIET(3) -- Milk (L y^(-1))
# DIET(4) -- Meat and poultry (kg y^(-1))
# DIET(5) -- Fish (kg y^(-1))
# DIET(6) -- Other seafood (kg y^(-1))
@Diet = (190, 64, 110, 95, 0, 0); # annual intakes as indicated

# Holdup times
# STOR_T(1) -- Fruit, non-leafy veg., grain (d)
# STOR_T(2) -- Leafy veg. (d)
# STOR_T(3) -- Meat (d)
# STOR_T(4) -- Milk (d)
# STOR_T(5) -- Fish (d)
# STOR_T(6) -- Crustacea, mollusks (d)
# STOR_T(7) -- Well water (d)
# STOR_T(8) -- Surface water (d)
# STOR_T(9) -- Livestock fodder (d)
@Storage = (14, 1, 1, 20, 7, 7, 1, 1, 45);     # holdup times as indicated

# Fractions of water supply coming from ground water (vs. surface water)
$FGW_DW = 1;  # Drinking water
$FGW_HH = 1;  # Household water
$FGW_LW = 1;  # Livestock water
$FGW_IR = 1;  # Irrigation water

# Crop and forage parameter arrays

# YV -- wet weight crop yields (kg m^(-2))
# YV(1) -- non-leafy veg.
# YV(2) -- leafy veg.
# YV(3) -- fodder
@Yield = (0.7, 1.5, 1.1);  # crop yields as indicated

# TE -- length of growing season (y)
# TE(1) -- non-leafy veg.
# TE(2) -- leafy veg.
# TE(3) -- fodder
@Growing_season = (0.17, 0.25, 0.08);   # growing seasons as indicated

# TIV -- translocation factor
# TIV(1) -- non-leafy veg.
# TIV(2) -- leafy veg.
# TIV(3) -- fodder
@Transloc_factor = (0.1, 1, 1);  # translocation factors as indicated
```

```
# RWET -- wet foliar interception fraction
# RWET(1) -- non-leafy veg.
# RWET(2) -- leafy veg.
# RWET(3) -- fodder
@Wet_intercept = (0.25, 0.25, 0.25);    # wet fol. interception fractions as
indicated

# RDRY -- dry foliar interception fraction
# RDRY(1) -- non-leafy veg.
# RDRY(2) -- leafy veg.
# RDRY(3) -- fodder
@Dry_intercept = (0.25, 0.25, 0.25);    # dry fol. interception fractions as
indicated

# Weathering constant for removal from plant surfaces (y^(-1))
$Lambda_weathering = 20;

# Transfer parameters for soil-to-plant and feed-to-animal-product are
# stored in a file in the \RESRAD directory (RFSAL.D34). To use Monte Carlo
# methods to vary these parameters, we have to rewrite these files at each
# iteration. The values used seem to be Bq kg^(-1) wet per Bq kg^(-1) dry
# soil. We vary the ones for Am, Pu, and Np according to NCRP Publication
# No. 129 Appendix D.
#
# NOTE: The Fmilk_GM value for Np in the RESRAD data base is 5E-6 d/kg;
# NCRP Report No. 129 gives 1E-5 with GSD 2.0. We retain the RESRAD value
%Biv_GM = ("Pu" => 1E-3, "Am" => 1E-3, "Np" => 0.02);
%Biv_GSD = ("Pu" => 2.5, "Am" => 2.5, "Np" => 2.5);
%Fmeat_GM = ("Pu" => 1E-4, "Am" => 5E-5, "Np" => 1E-3);
%Fmeat_GSD = ("Pu" => 1.5, "Am" => 1.5, "Np" => 2.0);
%Fmilk_GM = ("Pu" => 1E-6, "Am" => 2E-6, "Np" => 5E-6);
%Fmilk_GSD = ("Pu" => 1.6, "Am" => 2.0, "Np" => 2.0);
# Arrays for the sampled values
%Biv = ();
%Fmeat = ();
%Fmilk = ();

#-----------------------------------------------------------------------
# Scenario parameters
#-----------------------------------------------------------------------

# The following parameter controls the year for which the
# soil action levels are computed (it must correspond to one of the
# output times given below: 0 = 1971, 29 = 2000, etc.).
# At present, 2000 and 2100 are the only options.
$Scenario_date = 2000;     # see line 864 for reading summary.rep file -
changes need to be made

# Inhalation rate (m^3 y^(-1))
$Inhalation_rate = 10800;  # ICRP publication 23

# Drinking water intake (L y^(-1))
$DW_intake = 730;

# Soil ingestion rate (g y^(-1))
$Soil_ingestion_rate = 75;

# Fractions of time spent indoors and outdoors
$frac_indoors = 0.6;
$frac_outdoors = 0.4;
```

```
# Days each year spent on site
$days_on_site = 365;

# Building shielding factor for gamma rays
$Gamma_shield_factor = ($frac_outdoors + $frac_indoors * 0.5)
    * $days_on_site / 365;

# Dust reduction factor for indoor air
$Indoor_dust_factor = 0.7;

# Intake rates by livestock for contaminated materials:
$Fodder_meat = 68;  # kg d^(-1)
$Fodder_milk = 55;  # kg d^(-1)
$Water_meat = 0;    # L d^(-1)
$Water_milk = 0;    # L d^(-1)
$Livestock_soil = 0.5;     # kg d^(-1)

# Contaminated fractions of food and water sources
$CF_DW = 0;   # drinking water
$CF_HHW = 0;  # household water
$CF_LW = 0;   # livestock water
$CF_IW = 0;   # irrigation water
$CF_AQ = 0;   # aquatic food
$CF_plant = 1;      # dietary vegetables
$CF_meat = 1; # dietary meat
$CF_milk = 1; # milk


#-----------------------------------------------------------------------
# Output times
#-----------------------------------------------------------------------

# Note: in file DIMENSON.DAT in the \RESRAD directory,
# the variable NIY must be set equal to 5
# for this arrangement to work.
# T(1) = 0,
# T(2) = 29,
# T(3) = 1029,
# T(4) = 0,
# T(5) = 0,
# T(6) = 0,
# T(7) = 0,
# T(8) = 0,
# T(9) = 0,
# T(10) = 0,
@T = (0, 29, 1029, 0, 0, 0, 0, 0, 0, 0);       # output times (years after 1971)


#-----------------------------------------------------------------------
# Area factor data
#-----------------------------------------------------------------------

# The parameters $aä0, $bb0, $cc0 are interpolated from Table 4 of
# ANL/EAD/TM-82, Evaluation of the Area Factor Used in the
# RESRAD Code for the Estimation of Airborne Contaminant
# Concentrations of Finite Area Sources. The interpolation
# is for particle aerodynamic diameter 1 micrometer and
# the value of $wind_speed given below.

$WS_min = 3.7;
$WS_max = 4.3;
$Wind_speed = 0.5*($WS_min + $WS_max);  # average wind speed for Denver
```

```
# From Table 4 of ANL/EAD/TM-82 for particle diameter 1 micrometer:
@WS_tab = (1, 2, 5, 10);   # tabulated wind speed (m/s)
@aa = (1.9005, 1.6819, 0.7837, 0.1846);
@bb = (14.1136, 25.5076, 31.5283, 14.6689);
@cc = (-.2445, -.2278, -.2358, -.2627);
$Ntab = scalar @WS_tab;

$aa0 = linterp($Ntab, $Wind_speed, \@WS_tab, \@aa);
$bb0 = linterp($Ntab, $Wind_speed, \@WS_tab, \@bb);
$cc0 = linterp($Ntab, $Wind_speed, \@WS_tab, \@cc);

$New_area_factor = $aa0 / (1.0 + $bb0 * sqrt($Area)**$cc0);
$Old_area_factor = sqrt($Area) / (sqrt($Area)+$Dilution_length);
# Note: $Dilution_length was specified above and has a default
# value of 3.0 for the old methodology


#-----------------------------------------------------------------
# Save the current \RESRAD\DIMENSON.DAT file and substitute
# a copy that works for this problem.
#-----------------------------------------------------------------
@Dimenson = ();
open(DIMENSON, "<$RESRAD_path\\dimenson.dat") || die "Cannot open dimenson.dat
for input";
while ($line = <DIMENSON>)
{
    push @Dimenson, $line;
}
close DIMENSON;
open(DIMENSON, ">$RESRAD_path\\dimenson.dat") || die "Cannot open dimenson.dat
for output";
print DIMENSON <<END_DIMENSON;
 22  3  36  32  1  0  70  /NANUC,NIY,NPD,NPTS,NS,IHAFTIM,NPDS
 0  89  76  76  67  /NTAB(I,1),I=1,5
 125  89  76  76  67  /NTAB(I,2),I=1,5
END_DIMENSON
close DIMENSON;


#-----------------------------------------------------------------------
# Copy the RFSAL.* library from the \RESRAD directory for temporary storage
# in the working directory. These files contain dose conversion factors,
# soil-to-plant transfer factors, and feed-to-animal transfer factors.
#-----------------------------------------------------------------------
for $ext ("D34")     # for now we just work with .D34, but we can
                     # add other extensions if we want to do uncertainty
                     # on dose conversion factors
{
    system "copy $RESRAD_path\\RFSAL.$ext $Working_directory\\RFTMP.$ext >nul";
}

open(OUTDOSE, ">$outdose") || die "Cannot open output file $outdose";
print OUTDOSE "Scenario: $Title\n";
$tmp = ($Version eq "old")?"5.61":"5.82";
print OUTDOSE "RESRAD version $tmp\n";
print OUTDOSE "Total Annual Dose (mrem/year) for dates indicated:\n";
printf OUTDOSE "%d\t%d\t%s\t%s\n", $T[0]+1971, $T[1]+1971, "Fire",
    "Maximum";
open(OUTSAL, ">$outsal") || die "Cannot open output file $outsal";
print OUTSAL "Scenario: $Title\n";
print OUTSAL "RESRAD version $tmp\n";
```

```
print OUTSAL "Soil action levels for Rocky Flats radionuclides (pCi/g)\n";
print OUTSAL "Annual radiation dose limit: $Dose_limit (mrem/year)\n";
print OUTSAL "Effective year: $Scenario_date\n";
print OUTSAL "Am-241     \tNp-237     \tPu-238     \t";
print OUTSAL "Pu-239     \tPu-240     \tPu-241     \tPu-242\n";


#-------------------------------------------------------------------
# Monte Carlo loop
#-------------------------------------------------------------------

# Save default future output time (1029 years)
$T2sav = $T[2];

for $imc (1 .. $nmc)
{
RESTART:
    # Restore default future output time
    $T[2] = $T2sav;

    open(RADFILE, "<$radfile") || die "Cannot open rad file";
    open(XXFILE, ">$xxfile") || die "Cannot open output file";

    # Preliminary calculations
    #
    # (1) Estimate parameters related to leaching.
    #
       $Precip = sample("UNIFORM", $Precip_min, $Precip_max);             # m
y^(-1)
       $eps_eff = sample("UNIFORM", $eps_eff_min, $eps_eff_max);    #
dimensionless
       $eps_tot = sample("UNIFORM", $eps_tot_min, $eps_tot_max);

       # The following quantities are lognormal uncertainty
       # factors, with GM = 1, which multiply the nominal
       # estimates
       $Kd_Pu = sample("LNORM", $Kd_Pu_GM, $Kd_Pu_GSD);
       $Kd_Am = sample("LNORM", $Kd_Am_GM, $Kd_Am_GSD);
       $I = (1 - $Evap_transp) * ( (1 - $Runoff) * $Precip + $Irrig );
        $Rs = ($I / $Hydraulic_conductivity)**(1. /(2. * $b_parameter + 3.));
        $Leach_Pu = $I / ( $Delta_z • $eps_tot * $Rs * (1 + $Kd_Pu • $rho_b
            / $eps_tot * $Rs) );
#        print DUMP "Kd Pu: $Kd_Pu \n";
#        print DUMP "Kd Am: $Kd_Am \n";
#        print DUMP "Kd U: $Kd_U \n";
    # (2) Initial values in soil
    #
       # Sample the pCi g^(-1) of Pu-239+240 in the soil, using
       # the distribution parameters given above
       $Pu_239_240 = sample("LNORM", $Pu_239_240_GM, $Pu_239_240_GSD);

       # Use Pu leaching rate to adjust this contemporary (year 2000) value
       # to 1971:
       $Pu_239_240 *= exp($Leach_Pu * 29.0);

       # Use the 1971 normalized activities to separate Pu-239 and Pu-240
       $Act0{"Pu239"} = $ActNorm0{"Pu239"} / ( $ActNorm0{"Pu239"}
           + $ActNorm0{"Pu240"} ) * $Pu_239_240;
       $Act0{"Pu240"} = $Pu_239_240 - $Act0{"Pu239"};

       # Sample the 1971 Am-241:Pu-239+240 ratio and use it with
       # the Pu-239+240 soil concentration to calculate the
```

```
        # Am-241 soil concentration in 1971.
        $ratio = sample("LNORM", $Am_Pu_239_240_GM, $Am_Pu_239_240_GSD);
        $Act0{"Am241"} = $ratio * $ActNorm0{"Am241"}
            / ( $ActNorm0{"Pu239"}+$ActNorm0{"Pu240"} ) * $Pu_239_240;

        # Compute the 1971 Pu-241 soil concentration and make the
        # same random adjustment.
        $Act0{"Pu241"} = $ratio * $ActNorm0{"Pu241"}
            / ( $ActNorm0{"Pu239"}+$ActNorm0{"Pu240"} ) * $Pu_239_240;

        # Now compute the initial values of the remaining Pu isotopes.
        $Act0{"Pu238"} = $ActNorm0{"Pu238"}
            / ( $ActNorm0{"Pu239"}+$ActNorm0{"Pu240"} ) * $Pu_239_240;
        $Act0{"Pu242"} = $ActNorm0{"Pu242"}
            / ( $ActNorm0{"Pu239"}+$ActNorm0{"Pu240"} ) * $Pu_239_240;

    # (3) Sample the user-supplied air concentration and soil concentration
    #    for resuspension.

#***** Resuspension flux ratio calculation
#      If $flux_ratio is 1, no future flux due to a fire would exceed
#      the current vegetated flux, so we use the uncertainty (GSD)
#      derived from the regression based on contemporary sampling.
#      But if $flux_ratio exceeds 1, the flux due to some fire exceeds
#      the current vegetated flux, and we use the uncertainty (GSD)
#      corresponding to our estimate of soil flux following a fire,
#      which would likely produce a higher annual dose than the one
#      for contemporary vegetated conditions. The GM for a fire scenario
#      is the default air concentration multiplied by the simulated
#      flux ratio.

#((( -- After the above comment, delete everything down to the comment
#((( -- about tricking RESRAD and replace it with the following stuff:

    # Probability of a fire in any given year
    $p = sample("LOGUNIFORM", $pmin, $pmax);
    $Air_conc_2000 = sample("LNORM", $Air_conc_Pu_GM, $Air_conc_Pu_GSD);
    $flux_2000_0 = 5.38E-5;       # (mg m^(-2) s^(-1); ground-level flux for
            # year 2000 with normal ground cover
    $Delta_r0 = ($flux_ratio_0 - 1) * $flux_2000_0;
       # mg m^(-2) s^(-1); baseline incremental flux due to fire
    if ($Delta_r0 > 0)
    {
       $Delta_r_tmp = sample("LNORM", $Delta_r0, $Air_conc_Pu_fire_GSD);
       $flux_ratio = fire_flux($p, $Leach_Pu, $Delta_r_tmp);
    }
    else
    {
       $flux_ratio = 1;
    }
    if ($flux_ratio > 1)
    {
       # Compute RESRAD output time corresponding to year of fire
       $T[2] = 29 + $year_of_fire - 1;
       $Air_conc = $Air_conc_2000 * $flux_ratio;
    }
    else
    {
       $T[2] = 1029;
       $Air_conc = $Air_conc_2000;
    }
#(((( -- End replacement
```

```
        # Here is where we trick RESRAD:
        $ML_fact = $Air_conc / $Pu_in_soil_GM
            / $New_area_factor;    # mg m^(-3)

#(((( -- Add this:
        # In fire case, adjust $Pu_in_soil_GM factor for leaching
        if ($flux_ratio > 1) { $ML_fact *= exp(($year_of_fire - 1)
            * $Leach_Pu); }
#(((( -- End add this
#*****  End of resuspension flux ratio calculation

    # Possibly uncertain RESRAD parameters (pairwise independent).
    # First string is the variable name, exactly as it appears in the
    # RESRAD input file ($radfile below).
    @var = (
     [ "AREA", "CONST", $Area, 0 ],
     [ "WAREA", "CONST", $Watershed_area, 0 ],
     [ "COVER0", "CONST", 0, 0 ],        # Cover depth (m)
     [ "THICK0", "CONST", $Delta_z, 0 ],        # Thickness of contaminated zone
(m)
     [ "LCZPAQ", "CONST", 3000, 0 ],            # Length (m) parallel to aquifer
flow
     [ "HCCZ", "CONST", $Hydraulic_conductivity, 0 ],
     [ "BCZ", "CONST", $b_parameter, 0 ],       # "b parameter" for CZ
     [ "DENSAQ", "CONST", 1.8, 0 ],             # Density of saturated zone
(g/cm^3)
     [ "TPSZ", "CONST", 0.3, 0 ],        # Total porosity of SZ
     [ "EPSZ", "CONST", 0.1, 0 ],        # Effective porosity of SZ
     [ "HCSZ", "CONST", 44.5, 0 ],              # Hydraulic conductivity of SZ
     [ "HGWT", "CONST", 0.15, 0 ],              # Hydraulic gradient of SZ
     [ "VWT", "CONST", 0, 0 ],                  # Water table drop rate (m/y)
     [ "DWIBWT", "CONST", 10, 0 ],              # Well pump intake depth (m)
     [ "UW", "CONST", 250, 0 ],                 # Well pumping rate (m^3/y)
     [ "LM", "CONST", $Dilution_length, 0 ],
     [ "MODEL", "CONST", 0, 0 ],         # Nondispersion model of water
transport
     [ "NS", "CONST", 1, 0 ],                   # Number of layers in UZ
     [ "H(1)", "CONST", 3, 0 ],                 # Thickness of UZ (m)
     [ "DENSUZ(1)", "CONST", 1.8, 0 ],          # Density of UZ (g/cm^3)
     [ "TPUZ(1)", "CONST", 0.3, 0 ],            # Total porosity of UZ
     [ "EPUZ(1)", "CONST", 0.1, 0 ],            # Effective porosity of UZ
     [ "BUZ(1)", "CONST", 10.4, 0 ],            # "b parameter" for UZ
     [ "HCUZ(1)", "CONST", 44.5, 0 ],           # Hydraulic conductivity of UZ
     [ "DM", "CONST", $Mixing_depth, 0 ],       # Depth of mixing layer (m)
     [ "WIND", "UNIFORM", $WS_min, $WS_max],    # Wind speed (m/s)
     [ "DROOT", "CONST", $Root_zone, 0 ],       # Depth of root zone (m)
     [ "RI", "CONST", $Irrig, 0 ],              # Irrigation rate (m y^(-1))
     [ "IDITCH", "CONST", 0, 0 ],        # Irrigation mode (0 for overhead)
     [ "RUNOFF", "CONST", $Runoff, 0 ],         # Runoff coefficient
     [ "VCZ", "CONST", $Erode_CZ, 0 ],          # Erosion rate for contaminated
zone (y^(-1))
     [ "INHALR", "CONST", $Inhalation_rate, 0 ],
                                         # Inhalation rate (m^3 y^(-1))
     [ "ED", "CONST", 30, 0 ],                  # Exposure duration (y)
     [ "SHF1", "CONST", $Gamma_shield_factor, 0 ],
                                         # Building shielding for gamma rays
     [ "SHF3", "CONST", $Indoor_dust_factor, 0],
                                         # Dust reduction factor for indoors
     [ "SOIL", "CONST", $Soil_ingestion_rate, 0 ],
                                         # Soil ingestion rate (g y^(-1))
```


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```
     [ "DWI", "CONST", $DW_intake, 0 ],        # Drinking water intake (L y^(-
1))
     [ "FIND", "CONST", $frac_indoors, 0 ],    # Fraction of time indoors
     [ "FOTD", "CONST", $frac_outdoors, 0 ],   # Fraction of time outdoors
     [ "LFI5", "CONST", $Fodder_meat, 0 ],     # Livestock fodder intake for
meat (kg d^(-1))
     [ "LFI6", "CONST", $Fodder_milk, 0 ],     # Livestock fodder intake for
milk (kg d^(-1))
     [ "LWI5", "CONST", $Water_meat, 0 ],      # Livestock water intake for meat
(L d^(-1))
     [ "LWI6", "CONST", $Water_milk, 0 ],      # Livestock water intake for milk
(L d^(-1))
     [ "LSI", "CONST", $Livestock_soil, 0 ],   # Livestock soil intake (kg d^(-
1))
     [ "FGWDW", "CONST", $FGW_DW, 0 ],         # Frac. of drinking water from
groundwater
     [ "FGWHH", "CONST", $FGW_HH, 0 ],         # Frac. of household water from
groundwater
     [ "FGWLW", "CONST", $FGW_LW, 0 ],         # Frac. of livestock water from
groundwater
     [ "FGWIR", "CONST", $FGW_IR, 0 ],         # Frac. of irrigation water from
groundwater
     [ "FDW", "CONST", $CF_DW, 0 ],            # Contam. frac. of drinking water
     [ "FHHW", "CONST", $CF_HHW, 0 ],          # Contam. frac. of household
water
     [ "FLW", "CONST", $CF_LW, 0 ],            # Contam. frac. of livestock
water
     [ "FIRW", "CONST", $CF_IW, 0 ],           # Contam. frac. of irrigation
water
     [ "FR9", "CONST", $CF_AQ, 0 ],            # Contam. frac. of aquatic food
     [ "FPLANT", "CONST", $CF_plant, 0 ],      # Contam. frac. of plant food
     [ "FMEAT", "CONST", $CF_meat, 0 ],        # Contam. frac. of dietary meat
     [ "FMILK", "CONST", $CF_milk, 0 ],        # Contam. frac. of milk
     [ "EVAPTR", "CONST", $Evap_transp, 0 ],   # Evapotranspiration coefficient
     [ "WLAM", "CONST", $Lambda_weathering, 0 ],
                                          # Weathering rate for plant surfaces
(y^(-1))
     [ "DENSCZ", "CONST", $rho_b, 0 ],         # Bulk density of contaminated
zone (g cm^(-3))
     [ "BRDL", "CONST", $Dose_limit, 0 ] );    # Annual dose limit (mrem/year)
    $nvar = scalar @var;

    # Compare each variable name in the file with the list of uncertain
    # parameters in the array @var. Do a Monte Carlo sample for each hit,
    # or substitute for a "CONST" value when so indicated.

    LINELOOP:
    while ($line = <RADFILE>)
    {
       @fields = split /=/, $line;
       $vname = $fields[0];
       $vname =~ s/[ ]+//g;
       for $ivar (0 .. $nvar-1)
       {
           if ($vname eq $var[$ivar][0])
           {
              $dist = $var[$ivar][1];    # type of distribution

              # Generate a uniform random number
              do { $u = rand(1.0); } until $u > 0 && $u < 1;

              # Sample the distribution
```

```
            $value = sample( $dist, $var[$ivar][2], $var[$ivar][3] );

            $line = " $vname = $value,\n";
            goto PRINTLINE;
        }
    }
    if ($vname eq "TITLE")
    {
        $line = " TITLE = \'$Title\' ,\n";
        goto PRINTLINE;
    }
    if ($vname eq "DFFILE")
    {
        $line = " DFFILE = \'$dffile\' ,\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^T\([0-9]+\)/)
    {
        @part = split /[()]/, $vname;
        $line = " T($part[1]) = $T[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^DIET\([0-9]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " DIET($part[1]) = $Diet[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^STOR_T\([1-9]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " STOR_T($part[1]) = $Storage[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^YV\([1-3]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " $part[0]($part[1]) = $Yield[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^TE\([1-3]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " $part[0]($part[1]) = $Growing_season[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^TIV\([1-3]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " $part[0]($part[1]) = $Transloc_factor[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^RWET\([1-3]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " $part[0]($part[1]) = $Wet_intercept[$part[1]-1],\n";
        goto PRINTLINE;
    }
    if ($vname =~ /^RDRY\([1-3]\)/)
    {
        @part = split /[()]/, $vname;
        $line = " $part[0]($part[1]) = $Dry_intercept[$part[1]-1],\n";
```

```
            goto PRINTLINE;
        }

# NUCNAM = 'Ac-227', 'Am-241', 'Np-237', 'Pa-231', 'Pb-210', 'Pu-238',
# 'Pu-239', 'Pu-240', 'Pu-241', 'Pu-241', 'Pu-242', 'Ra-226',
# 'Ra-228', 'Th-228', 'Th-229', 'Th-230', 'Th-232', 'U-233', 'U-234',
# 'U-235', 'U-236', 'U-238', 'LAST',
# S = 0, .3, 3*0, 1, 8.42, 1.58, 2*40, 1, 11*0,
# W = 22*0,
# DCACTC = 2*20,-1, 50, 100, 6*2000, 2*70, 4*60000, 5*50,
# DCACTU1 = 2*20,-1, 50, 100, 6*2000, 2*70, 4*60000, 5*50,
# DCACTS = 2*20,-1, 50, 100, 6*2000, 2*70, 4*60000, 5*50,
# RLEACH = 22*0,
# SOLUBK0 = 22*0,
        if ($vname eq "DCACTC" || $vname eq "DCACTU1" || $vname eq "DCACTS" )
        {
        # K_d values for contaminated zone, unsaturaded zone, and
        # saturated zone
            $line = " $vname = 20, $Kd_Am, -1, 50, 100,\n";
            $line = $line." 6*$Kd_Pu, 2*70, 4*60000, 5*50,\n";
            goto PRINTLINE;
        }
        if ($vname eq "S")  # Nuclide activities (pCi/g)
        {
            # Note: 1E-10 is a negligible nonzero initial value for
            # the activity of Np-237 to trigger RESRAD to compute
            # a soil action level for this nuclide
            $line = " S = 0, $Act0{\"Am241\"}, 1E-10, 2*0 ,\n";
            $line = $line." $Act0{\"Pu238\"}, $Act0{\"Pu239\"},\n";
            $line = $line." $Act0{\"Pu240\"}, $Act0{\"Pu241\"},\n";
            $line = $line." $Act0{\"Pu241\"}, $Act0{\"Pu242\"}, 11*0 ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "MLINH")     # mass-loading for inhalation
        {
            $line = " MLINH = $ML_fact ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "MLFD")      # mass-loading for foliar deposition
        {
            $line = " MLFD = $ML_fact ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "PRECIP")
        {
            $line = " PRECIP = $Precip ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "TPCZ")
        {
            $line = " TPCZ = $eps_tot ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "EPCZ")
        {
            $line = " EPCZ = $eps_eff ,\n";
            goto PRINTLINE;
        }
        if ($vname eq "SELPATH")
        {
            $line = " SELPATH = $select_path ,\n";
            goto PRINTLINE;
```

```
      }

      PRINTLINE:
      print XXFILE "$line"; # $line may be the one read in or an alteration
   }
   close RADFILE;
   close XXFILE;


   # Sample transfer parameters for Pu, Am, and Np
   # and make formatted string representations of their values
   for $elem ("Pu", "Am", "Np")
   {
      $Biv{$elem} = sprintf "%.1E",
          sample("LNORM", $Biv_GM{$elem}, $Biv_GSD{$elem});
      $Biv{$elem} =~ s/E([-+])00([0-9])/E$1$2/g;
      $Fmeat{$elem} = sprintf "%.1E",
          sample("LNORM", $Fmeat_GM{$elem}, $Fmeat_GSD{$elem});
      $Fmeat{$elem} =~ s/E([-+])00([0-9])/E$1$2/g;
      $Fmilk{$elem} = sprintf "%.1E",
          sample("LNORM", $Fmilk_GM{$elem}, $Fmilk_GSD{$elem});
      $Fmilk{$elem} =~ s/E([-+])00([0-9])/E$1$2/g;
   }

   # Make a script file for a call to the batch editor sed.
   # The script will direct sed to read the file RFSAL.D34 and
   # replace the transfer factors on the lines beginning with
   # "Pu", "Am", and "Np" with the sampled values just computed.
   open (SCRIPT, ">$Working_directory\\script")
      || die "Cannot open script file for sed";
   print SCRIPT <<END_SCRIPT;
/^Pu/c\\
Pu         $Biv{"Pu"}   $Fmeat{"Pu"}   $Fmilk{"Pu"}
/^Am/c\\
Am         $Biv{"Am"}   $Fmeat{"Am"}   $Fmilk{"Am"}
/^Np/c\\
Np         $Biv{"Np"}   $Fmeat{"Np"}   $Fmilk{"Np"}
END_SCRIPT

   close SCRIPT;

   # Now have the system call sed and point it to the script file
   # and RFSAL.D34 in the \RESRAD directory. The necessary
   # command string is long, so we do it in several pieces.
   $cmd = "sed -f $Working_directory\\script";       # first piece
   $cmd .= " $Working_directory\\RFTMP.D34";  # second piece
   $cmd .= " >$RESRAD_path\\RFSAL.D34";       # tail
   system "$cmd";  # system command

   #-----------------------------------------------------------------
   # Move $xxfile to \resrad and execute RESMAIN3
   #-----------------------------------------------------------------

   system "copy $xxfile $RESRAD_path\\$xxfile >nul"; # suppress DOS copy
message
   die "Cannot cd to $RESRAD_path" unless chdir "$RESRAD_path";
   system "resmain3 $xxfile 0 >nul" || die "Cannot execute resmain3";

   system "erase $xxfile" || print "Cannot erase $xxfile from $RESRAD_path";
   die "Cannot cd to home directory" unless
      chdir "$Working_directory";
```

```
#-----------------------------------------------------------------
# Get the radionuclide concentrations in the contaminated zone
# for each output year and store them in the hash array %conc
# for later use.
#-----------------------------------------------------------------
%conc = ();
open(CONCENT, "<$RESRAD_path\\concent.rep")
   || die ("Could not open concent.rep");
$found = 0;
for $count (0 .. 2)
{
   while ($line = <CONCENT>)
   {
       if ($line =~ /Concentration of radionuclides in environmental media/)
       {
          $found = 1;
          last;
       }
   }
   for (1 .. 7) { $line = <CONCENT>; }     # skip next 7 lines

   # Iterate over the tabular list, one line for each radionuclide.
   # This list contains not only our principal radionuclides, but also
   # all descendent species, which we wish to exclude (because RESRAD
   # will implicitly include these descendents in its calculations). Thus,
   # we must look at each line and see whether the nuclide name in the
   # first field matches some name in the array @nuc_name. If so,
   # the line corresponds to a principal radionuclide, and we store
   # the number in the second field, which is the nuclide concentration
   # in the contaminated zone. Otherwise, we move to the next line.
   # A line with the pattern "-------" marks the end of the table.
   while ($line = <CONCENT>)
   {
       last if ($line =~ /-------/);
       chomp $line;
       $line =~ s/^[ ]+//;    # delete initial spaces
       @field = split /[ \t]+/, $line;
       $name = $field[0];
       $found = 0;

       # Is this nuclide in the list of principal nuclides?
       foreach (@nuc_name)
       {
          if ($_ ne $name) { $found = 1; last; }
       }
       if ($found == 0) { next; }    # not in list
       else     # yes, principal radionuclide; put in hash array
       {
          $conc{$name}[$count] = $field[1];
       }
   }
}
close CONCENT;

#-----------------------------------------------------------------
# Recover the dose estimates at times 0, 29, and $T[2] y,
# corresponding to 1971, 2000, and date of the fire.
#-----------------------------------------------------------------
open(SUMMARY, "<$RESRAD_path\\summary.rep")
   || die ("Could not open summary.rep");
# Look for the phrase "Total Dose TDOSE", which occurs only one place
# in the file
```

```
    $found = 0;
    while ($line = <SUMMARY>)
    {
       if ($line =~ /Total Dose TDOSE/) { $found = 1; last; }
    }
    if ($found == 0)
    {
       close SUMMARY;
       goto RESTART;
    }

    # BEGIN replacement
    for (1 .. 4) { $line = <SUMMARY>; } # skip 3 lines and retain the 4th
    chomp $line;            # remove \n character from end of line
    $line =~ s/^[ ]+//;     # remove initial space characters
    @field = split /[ \t]+/, $line;
    for (1 .. 2) { shift @field; }      # remove "t (years):"
    @dosedate = ();
    for $f (@field)
    {
       push @dosedate, $f+1971;
    }
    $line = <SUMMARY>;      # get "TDOSE(t):" line
    chomp $line;
    $line =~ s/^[ ]+//;     # remove initial space characters
    @field = split /[ \t]+/, $line;
                         # split the line at spaces or tabs
                         # and put the fields in an array.
    shift @field;           # remove a label field "TDOSE(t):"

    # For the maximum dose, we use either (a) the maximum of the tabulated
    # values at or beyond the scenario date, or (b) the maximum dose
    # calculated by RESRAD if the calculated time is >= the scenario date.
    #
    # First set up default case (a)
    $dosemax = 0;
    for $i (0 .. (scalar @field)-1)
    {
       # Change awkward formatting such as 2.43E+003 to 2.43E+3
       $field[$i] = sprintf "%.3G", $field[$i];
       $field[$i] =~ s/E[+]00/E+/;
       printf OUTDOSE "%s\t", $field[$i];
       if ($dosedate[$i] >= $Scenario_date)
       {
           if ($dosemax < $field[$i]) { $dosemax = $field[$i]; }
       }
    }
    for (1 .. 2) { $line = <SUMMARY>; }
    @field = split /[ \t]+/, $line;
    $dmax = $field[2];
    $tmax = $field[7]+1971;
    # Change awkward formatting such as 2.43E+003 to 2.43E+3
    $dosemax = $dmax if ($tmax >= $Scenario_date);    # case (b)
    $dosemax = sprintf "%.3G", $dosemax;
    $dosemax =~ s/E[+]00/E+/;
    printf OUTDOSE "%s", $dosemax;
#***** Modified to show fire/no-fire information at end of line
    $fire_date = 2000+$year_of_fire;
    if ($flux_ratio > 1) { printf OUTDOSE "\tF %d
%.2G\n",$fire_date,$flux_ratio; }
    else { printf OUTDOSE "\tN 2000 %.2G\n",$flux_ratio; }
#***** End mod
```

```
    #-----------------------------------------------------------------
    # For each radionuclide, determine the fraction of the total dose
    # that is from inhalation and plant ingestion (resuspension
    # pathways).
    #-----------------------------------------------------------------
    %resusp_frac = ();
    # This search finds year t = 0 (1971)
    while ($line = <SUMMARY>)
    {
       last if ($line =~ /Water Independent Pathways/);
    }
    for $i (0 .. 1)
    {
       # This one finds t = 29 (2000) for $i = 0 and
       # the fire year for $i = 1
       while ($line = <SUMMARY>)
       {
           last if ($line =~ /Water Independent Pathways/)
       }
       for (1 .. 4) { $line = <SUMMARY>; } # skip down to tabular lines
       for (1 .. 7) # iterate over radionuclides in the table
       {
           $line = <SUMMARY>;
           $line =~ s/^[ ]+//;    # delete initial spaces
           @field = split /[ \t]+/, $line;
           if ($i == 0) { push @nuc_name, $field[0]; }
            # $field[2] is the fraction of the dose due to ground
            # $field[4] is the fraction of the dose due to inhalation,
           # $field[8] is the fraction from plant ingestion,
           # $field[10] is the fraction from local meat,
           # $field[12] is the fraction from local milk.
            # $field[14] is the fraction from soil ingestion.
           # These pathways are directly fed by resuspension. In the
           # case of plants (produce), nearly all of the dose comes
           # from foliar deposition rather than root uptake. For
           # meat and milk, a fraction is from the animal's ingestion
           # of soil, but we neglect this. The remaining water-independent
           # pathways are external dose and direct ingestion of
           # contaminated soil. Water-dependent pathways are included
           # in the complement, but these are assumed to be negligibly
           # dependent on resuspension.
           $resusp_frac{$field[0]}[$i] = $field[4] + $field[8] + $field[10]
              + $field[12];
#          print STDERR "$field[0]: $field[4] $field[8]\n";

        }

# get total fractions of dose for year 2000 if $flux_ratio<=1, fire year
# if $fluc_ratio>1
        $line=<SUMMARY>;
        $line=<SUMMARY>;
        $line =~ s/^[ ]+//; # delete initial spaces
        @field = split /[ \t]+/, $line;
        if  ($flux_ratio>1 && $i == 1)
        {
          print DUMP "$i  $field[2]  $field[4]  $field[8]  $field[10]
$field[12]  $field[14]";
        }
        if ($flux_ratio<=1 && $i == 0)
        {
```

```
          print DUMP "$i  $field[2]  $field[4]  $field[8]  $field[10]
$field[12]  $field[14]";
        }
    }

    #-----------------------------------------------------------------
    # Now search further in the file to locate the RESRAD-calculated
    # SAL values for the principal radionuclides. These are the values
    # determined by the 15 mrem maximum annual dose criterion.
    #-----------------------------------------------------------------
    $found = 0;
    while ($line = <SUMMARY>)
    {
       if ($line =~ /Single Radionuclide Soil Guidelines/)
       {
           $found = 1;
           last;
       }
    }
    die "Pattern /Single Radionuclide Soil Guidelines/ not found"
       if ($found == 0);
    for (1..4) { $line = <SUMMARY>; }   # skip 4 lines
    %sal_array = ();       # storage array
    for $i (1..7)   # iterate over the 7 lines of the Guidelines table
    {
       $line = <SUMMARY>;
       chomp $line;
       $line =~ s/^[ ]+//; # remove initial space characters
       $line =~ s/[*]//g;  # remove asterisks
       @field = split /[ \t]+/, $line;
                          # split the line at spaces or tabs
                          # and put the fields in an array.
       $name = $field[0];
       shift @field; # shift past radionuclide name
       shift @field; # discard 1971 value

        $sal = $field[0];       # this is the value for 2000
       if ($flux_ratio > 1) { $sal = $field[1]; }

       # Adjust soil action level from 1971 to 2000 as baseline year
       $sal *= $conc{$name}[1] / $conc{$name}[0];
       printf OUTSAL "%.2E", $sal;
       if ($i < 7) { print OUTSAL "\t"; }
       else { print OUTSAL "\n"; }
       $sal_array{$name} = $sal;  # store SAL for possible later use
    }
    # END replacement

    close SUMMARY;
    if ($imc % 5 == 0) { print STDERR "$imc"; }
    else { print STDERR ".";}

#-----------------------------------------------------------------
# End Monte Carlo loop
#-----------------------------------------------------------------
}

close OUTDOSE;
close OUTSAL;
# copy summary.rep to current directory
   system "del $Working_directory\\summary.rep >nul";
   system "del $Working_directory\\concent.rep >nul";
```

```
   system "copy c:\\resrad\\summary.rep >nul";
   system "copy c:\\resrad\\concent.rep >nul";

#-----------------------------------------------------------------
# Restore DIMENSON.DAT file in directory \RESRAD
#-----------------------------------------------------------------
open(DIMENSON, ">$RESRAD_path\\DIMENSON.DAT")
    || die "Cannot open DIMENSON.DAT for restoration";
for $d (@Dimenson)
{
    print DIMENSON "$d";
}
close DIMENSON;

#-----------------------------------------------------------------
# Restore RFSAL.* library files in directory \RESRAD
#-----------------------------------------------------------------
for $ext ("D34")
{
    system "copy $Working_directory\\RFTMP.$ext $RESRAD_path\\RFSAL.$ext >nul";
    system "erase $Working_directory\\RFTMP.$ext >nul";
}

system "copy $RESRAD_path\\summary.rep $Working_directory\\summary.rep"
    if ($nmc == 1);

print STDERR "\nDone\n";
exit;

#-----------------------------------------------------------------
# End of main program
#-----------------------------------------------------------------

# Front-end subroutine for Monte Carlo sampling.
# Calling sequence:
# $value = sample("LNORM", $GM, $GSD);
sub sample
{
    local ($type, $p1, $p2) = @_; # arguments: distribution and 2 parameters
    local ($u, $rv);
    if ($type eq "CONST")
    {
       $rv = $p1;
    }
    elsif ($nmc <= 1)
    {
       if ($type eq "UNIFORM") { $rv = 0.5*($p1+$p2); }
       elsif ($type eq "LOGUNIFORM") { $rv = sqrt($p1*$p2); }
       else { $rv = $p1; }
    }
    else
    {
       # Get a uniformly distributed [0,1] random number
       do { $u = rand(1.0) } until $u>0 && $u<1;      # discard 0 and 1
       SWITCH2:
       {
       $rv = NORM_sample($p1, $p2, $u), last SWITCH2 if ($type eq "NORM");
       $rv = LNORM_sample($p1, $p2, $u), last SWITCH2 if ($type eq "LNORM");
       $rv = UNIFORM_sample($p1, $p2, $u),
           last SWITCH2 if ($type eq "UNIFORM");
       $rv = LOGUNIFORM_sample($p1, $p2, $u),
           last SWITCH2 if ($type eq "LOGUNIFORM");
```

```
      die "Distribution type not found in subroutine sample";
      }
    }
    return ($rv);
}

sub linterp
{
    local ($N = shift);
    local ($x = shift);
    local ($pxtab, $pytab) = @_;

    local ($i, $found = 0);
    for $i (0 .. $N-2)
    {
      local ($x0 = $$pxtab[$i]);
      local ($x1 = $$pxtab[$i+1]);
      next if (!($x >= $x0 && $x < $x1));

      local ($y0 = $$pytab[$i]);
      local ($y1 = $$pytab[$i+1]);
      local ($y = $y0 + ($y1-$y0)/($x1-$x0)*($x-$x0));
      $found = 1;
      last;
    }
    if (!$found)
    {
      if ($x >= $$pxtab[$N-1]) { return $$pytab[$N-1]; }
      if ($x <= $$pxtab[0]) { return $$pytab[0]; }
    }
    else
    {
      return $y;
    }
}


#-----------------------------------------------------------------
# Subroutine %fire_flux estimates a flux ratio: fire / no-fire.
# If the returned ratio is 1, we assume no fire and use the flux
# for year 2000 ground cover.
#-----------------------------------------------------------------
# Arguments:
#     p = probability of a fire in any specified year.
# alpha = decay-rate coefficient for Pu radioactivity in the soil.
#    r0 = worst-case estimate of flux ratios. This would be the value for
#         a fire that occurred in the first year considered, with
#         unvegetated soil that persisted for the entire year.
# Internal random variables:
#     f = random modifying factor to adjust for fire area
#     g = random modifying factor to allow for such influences as the
#         time of year the fire occurs and the distance of the fire's
#         location from the scenario subject

#((( -- Replace the entire fire_flux subroutine with the version below:
sub fire_flux
{
    # Parameters
    local $p = $_[0];
    local $alpha = $_[1];
    local $Delta_r = $_[2];
```

```
    # Other variables
    local $ratio;
    $year_of_fire = 1;     # global variable to be calculated
    local ($N = 1000, $M = scalar @Burn_area);
    local $f, $g;   # random modifying factors
    local $m;       # random index for burn area
    local $i;
    local $u;
    local $rtemp;
    local $chi_star;

    $ratio = 1;
    for ($i = 1; $i <= $N; $i++)
    {
       $u = rand(1.0);
       next if ($u > $p);  # no fire in year $i -- proceed to next year

       # Otherwise, there is a fire. Compute modifying factor $f,
       # which is the ratio of a random burn area based on regional
       # statistics to the area at risk on the RF site:

       $m = int rand($nfs);      # random index for @firestream array
       $m = $nfs-1 if ($m >= $nfs);
       $chi_star = $firestream[$m];
           # concentration (Bq m^(-3)) per unit flux from the circular
           # burn area (mg m^(-2) s^(-1))
       $u = rand(1.0);
       $g = 0.5 * $u + (1.0 - $u);
       $rtemp = (1.0 + $chi_star * $Delta_r / $Air_conc_2000)
           * $g * exp( -$alpha * ($i-1) );
       if ($rtemp > $ratio)
       {
           $ratio = $rtemp;
           $year_of_fire = $i;    # set global variable
       }
    }
    $ratio = 1 if ($ratio < 1);
    return $ratio;
}
#((( -- End fire_flux replacement
```

DO_GRAPH.PL

```
#!/usr/bin/perl -w

# usage for ASR
# usage perl do_graph.pl mc
# the files mcsal.dat, which is a copy of mcsal.prt must be in the
# default directory
# end usage ASR 09/27/99

die "Usage: do_graph.pl name_field" if (scalar @ARGV != 1);
$name = $ARGV[0];
if (-e "${name}graph.dat") { system "erase ${name}graph.dat"; }
$N = 50;      # 1 + number of points plotted per logarithmic decade
$factor = exp( log(10) / $N );
$Pu = 1 / $factor;
do
{
BEGIN_LOOP:
    $Pu *= $factor;
    $cmd = "type ${name}sal.dat|perl sratio.pl $Pu|perl cumprob.pl 1.0";
    $cmd .= "|awk \"\{print 1-\$1\}\"|";
    open (FEEDBACK, $cmd) || die "Cannot open FEEDBACK";
    $_ = <FEEDBACK>;
    chomp $_;
    $p = $_;
    goto BEGIN_LOOP if ($p <= 0);
    system "awk \"BEGIN {print $Pu,$p}\" >>${name}graph.dat";
} until $Pu >= 10000 || $p == 1;

open(TEMP, "<${name}graph.dat") || die "Could not open ${name}graph.dat for
input";
@x = ();
@y = ();
GET_TEMP:
while ($line = <TEMP>)
{
    chomp $line;
    $line =~ s/^[ ]+//;    # remove initial spaces
    @field = split /[ \t]+/, $line;
    $field[1] =~ s/[ ]+//g;
    next GET_TEMP if ($field[0] <= 0 || $field[1] <= 0 || $field[0] eq "0"
       || $field[1] eq "0");
    push @x, $field[0];
    push @y, $field[1];
}
close TEMP;
$nx = scalar @x;

# Tabulate x-values corresponding to 5%, 10%, 20%, and 50%
@p0 = (.05, .1, .2, .5);
@x0 = ();
$np = scalar @p0;
for $i (0 .. $np-1)
{
    $p = $p0[$i];
    JCRIT:
    for $jcrit (0 .. $nx-1)
    {
       $j = $jcrit;
       last JCRIT if ($y[$jcrit] > $p);
    }
```

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```
    $xi = $x[$j-1] + ($x[$j]-$x[$j-1])/($y[$j]-$y[$j-1])
       * ($p - $y[$j-1]);
    push @x0, $xi;
}
open (TEMP, ">${name}table.txt") || die "Cannot open ${name}table.txt for
output";
for $i (0 .. $np-1)
{
    printf TEMP "%.0f %%\t%G\n", $p0[$i] * 100, $x0[$i];
}
close TEMP;

# Determine plot area for graph
JMIN:
for $j (0 .. $nx-1)
{
    $jmin = $j;
    last JMIN if ($y[$j] >= 0.01);
}
$jmin--;
$jmin = 0 if ($jmin < 0);
$xmin = $x[$jmin];
$ymin = 0.01;
JMAX:
for ($jmax = $nx-1; $jmax >= $jmin+1; $jmax--)
{
    last JMAX if ($y[$jmax] < 1);
}
$jmax++ unless $jmax == $nx-1;
$xmax = $x[$jmax];
$ymax = 1;
$xmin_power = int( log($xmin)/log(10) );
if (10**$xmin_power > $xmin) { $xmin_power--; }
#$ymin_power = int( log($ymin)/log(10) );
#if (10**$ymin_power > $ymin) { $ymin_power--; }
$xmax_power = int( log($xmax)/log(10) );
if (10**$xmax_power < $xmax) { $xmax_power++; }
#$ymax_power = int( log($ymax)/log(10) );
#if (10**$ymax_power < $ymax && $ymax <= 1) { $ymax_power++; }
$xmin = 10**$xmin_power;
$xmax = 10**$xmax_power;

# Write a jgraph script for plotting the graph
open(GRAPH, ">".${name}."graph.jgr") || die "Could not open $name.jgr for
output";
print GRAPH <<ENDGRAPH;
newgraph
border clip
x_translate 1
y_translate 1
xaxis min $xmin max $xmax log hash_labels font Helvetica
label font Helvetica-Bold : Plutonium-239+240 (pCi/g)
yaxis min $ymin max $ymax log hash_format G hash_labels font Helvetica
label font Helvetica-Bold : Probability of exceeding annual dose limit

newline pts shell : awk \"{print \$1,\$2}\" ${name}graph.dat
ENDGRAPH
close GRAPH;

# Run jgraph to compute the PostScript file of the graph
# *** For Art's version, comment this out
#system "jgraph <${name}graph.jgr >${name}graph.ps";
```

```
exit;
```

CUMPROB.PL

```
#!/usr/bin/perl -w
# Usage: cumprob.pl x <dist_file >value of CDF at x

die "Usage: cumprob.pl x <dist_file" if (scalar @ARGV != 1);
$x = $ARGV[0];
$max_less = -1E90;
$min_greater = 1E90;
$N_less = $N_greater = 0;
@xi0 = ();
while (<STDIN>)
{
    chomp $_;
    $_ =~ s/^[ ]+//;       # lose initial spaces
    @field = split /[ \t]+/, $_;
    push @xi0, $field[0];
}
$nxi = scalar @xi0;
@xi1 = sort { $a <=> $b } @xi0;
$ximax = $xi1[$nxi-1];
$ximin = $xi1[0];
for $xi (@xi1)
{
    if ($xi == $x)
    {
       if ($xi == $ximax)
       {
           $N_less++;
       }
       elsif ($xi == $ximin)
       {
           $N_greater++;
       }
       else
       {
           $N_less += 0.5;
           $N_greater += 0.5;
       }
       $max_less = $min_greater = $xi;
    }
    elsif ($xi < $x)
    {
       $N_less++;
       $max_less = $xi if ($xi > $max_less);
    }
    else
    {
       $N_greater++;
        $min_greater = $xi if ($xi < $min_greater);
    }
}
if ($max_less == $min_greater)
{
    $prob = $N_less / ($N_less + $N_greater);
}
elsif ($max_less == -1E90 && $min_greater < 1E90) { $prob = 0; }
elsif ($max_less > -1E90  && $min_greater == 1E90) { $prob = 1; }
elsif ($max_less == -1E90 && $min_greater == 1E90) { $prob = -1; } #error
else   #interpolate
{
    $p1 = $N_less / ($N_less + $N_greater);
```

```
    $p2 = ($N_less+1) / ($N_less + $N_greater);
    $m1 = $max_less;
    $m2 = $min_greater;
    $prob = $p1 + ($p2 - $p1) / ($m2 - $m1) * ($x - $m1);
}
print "$prob\n";
exit;
```

SRATIO.PL

```
#!/usr/bin/perl -w
# sratio.pl calculates the distribution of the sum of ratios given
# an estimate of Pu-239+240 in the soil, using ratios of the isotopes
# in the year 2000 and the distributions of soil action levels for
# the several isotopes.
#
# Usage: sratio.pl Pu-239+240 in pCi/g <SAL_table >S_ratio_distribution

die "Usage: sratio.pl Pu-239+240 in pCi/g <SAL_table >S_ratio_dist"
    if ((scalar @ARGV) != 1);

# The following data are isotope-specific ratios relative to 1 pCi
# of Pu-239+240 in the soil in the year 2000:
%isorat = ( "Am-241" => .11123, "Np-237" => 7.855E-7, "Pu-238" => .01319,
    "Pu-239" => .8428, "Pu-240" => .1572, "Pu-241" => .7980,
    "Pu-242" => 7.616E-6 );

$Pu_239_240 = $ARGV[0];
for $i (1 .. 6) { $line = <STDIN>; }
chomp $line;
@isoname = split /[ \t]+/, $line;
$niso = scalar @isoname;
while ($line = <STDIN>)
{
    chomp $line;
    $line =~ s/^[ ]+//;    # lose any initial spaces
    @field = split /[ \t]+/, $line;
    last if (scalar @field < $niso);
    $sratio = 0;
    for $i (0 .. $niso-1)
    {
       $name = $isoname[$i];
       $name =~ s/[ \t]//;
       $sratio += $Pu_239_240 * $isorat{$name} / $field[$i];
    }
    die "sratio is zero" if ($sratio == 0);
    print "$sratio\n";
}
exit;
```

APPENDIX B

ALTERNATIVE GROUNDWATER DOSE CALCULATION FROM MEASUREMENTS BY LITAOR

APPENDIX B

ALTERNATIVE GROUNDWATER DOSE CALCULATION FROM MEASUREMENTS BY LITAOR

B.1 INTRODUCTION

The groundwater pathway for plutonium isotopes used in this radionuclide soil action level (RSAL) analysis begins in the year 1971. Contaminant concentrations in the aquifer are initially at zero and become contaminated only after leachate from the contaminated zone reaches the aquifer. Using the RESRAD model, contaminant travel times from the contaminated zone to the aquifer were greater than 1000 years for the plutonium and americium isotopes. Because the maximum dose for all scenarios occurs in the first year of exposure (year 2000), doses from groundwater sources are zero for all scenarios. As explained in Section B.2.2, adding the groundwater pathway with the 1 m y^{-1} irrigation rate actually decreases the RSALs because the additional water from irrigation depletes radionuclides in the contaminant zone faster than without the water.

Unpublished measurements made by I. Litaor et al (1999), however, have indicated plutonium and americium are already present in the groundwater underlying the Rocky Flats Plant (RFP). Transport mechanisms not considered in the RESRAD model (and most groundwater transport models) are suspected to be the driving force behind the migration of plutonium to the groundwater. Therefore, it is not possible given the time and budget constraints of this project to incorporate these processes in the RSAL calculation. Nevertheless, we believe some assessment of the groundwater pathway should be made using the data compiled by Litaor. Appendix B provides such an assessment. We attempt to put the groundwater pathway into perspective by computing ingestion doses assuming the *Risk Assessment Corporation* (*RAC*) exposure scenarios and the measured concentrations. These results provide some measure of the potential impact of groundwater doses on the RSALs.

B.2 PLUTONIUM AND AMERICIUM MEASUREMENTS IN GROUNDWATER

Litaor and colleagues have been studying the movement of plutonium in soil around the RFP over the last 5 years (Litaor et al. 1998, 1996, 1995, 1994; Litaor and Ibrahim 1996). In an unpublished draft of his work, he reports ^{239}Pu and ^{241}Am concentrations in interstitial pore water, runoff, and seep water. These measurements were made during the spring and early summer of 1995, where in surface soils, normally unsaturated and aerobic conditions became saturated and probably anaerobic during an unusually wet period covering 65 days. A calibrated numeric groundwater model suggested that 103,000 m^3 of water was discharged through the outflow boundary. The numeric simulation agreed extremely well with measured effluent discharge from a holding pond that presumably collected all the upslope groundwater flow.

Litaor observed that most of the plutonium (~90%) in pore water was associated with colloids (0.1 nm to 0.45 μm) and larger particles (>0.45 μm). However, as chemical conditions changed to anaerobic, more plutonium (~25%) was associated with the dissolved phase (particles <1 nm). It was suspected that prolonged saturated conditions coupled with reducing conditions led to enhanced dissolved phase actinide migration for a brief period of time.

The groundwater model in RESRAD does not account for colloidal transport, nor is it capable of incorporating geochemical changes or transient infiltration events such as those that occurred in the spring and early summer of 1995. To account for these deficiencies in the RESRAD model, we have included this alternative groundwater assessment. This assessment uses the data measured Litaor, combined with drinking water ingestion rates and ICRP Publication 70 dose conversion factors to calculate the doses one would have received had they been drinking water discharged from the site. We do not attempt to include this assessment in the RSAL calculation. To do so would require a complete reevaluation of the scenarios and other factors that affect action levels. These dose calculations are presented here to (a) address dose impacts from the site as it currently stands and (b) address deficiencies in the RESRAD groundwater model.

B.2.1 Conceptual Model

Litaor measured concentrations in interstitial pore water in eight soil horizons ranging down to a depth of 6.6 m and also in seep water and surface runoff water (Table B-1). We do not believe its a credible scenario for someone to consume interstitial pore water near the surface (<3 m) because most of the time these soils are unsaturated and, therefore, it would make little sense to drill a well into these layers. Below this depth, water tends to be present in places year round, depending on location. A scenario where someone drinks surface runoff is also not considered to be credible because this would most certainly be an unreliable water source. The scenarios considered then involve water consumption from three sources: a well drilled down to a depth >3 m, seep water, and discharge water from the study area.

To compute the concentration in water discharged from the study area, we divided the plutonium and americium flux estimated to have been released from the area (24 and 4.3 MBq, respectively) by the total amount of water discharged during the measurement period (1×10^5 m^3). This calculation results in water concentrations of 0.0087 pCi L^{-1} and 0.0015 pCi L^{-1} for $^{239/240}Pu$ and ^{241}Am, respectively.

Table B-1. Concentrations of $^{239/240}Pu$ and ^{241}Am in Pore Water, Seep water, and Runoff as Measured by Litaor[a]

Depth (m)	Mean $^{239/240}Pu$ (pCi L^{-1})	Min $^{239/240}Pu$ (pCi L^{-1})	Max $^{239/240}Pu$ (pCi L^{-1})	Mean ^{241}Am (pCi L^{-1})	Min ^{241}Am (pCi L^{-1})	Max ^{241}Am (pCi L^{-1})
<0.2	99.9	1.48	877	18.5	0.137	178
0.2-0.4	15.2	0.133	287	2.22	0.022	40.7
0.4-0.7	5.55	0.030	62.9	0.888	0.001	9.62
0.7-1.5	0.851	0.019	3.40	0.115	0.011	0.444
1.5-2.0	0.255	0.067	0.999	0.052	0.007	0.130
2.0-2.7	0.270	0.003	1.33	0.041	0.001	0.115
2.7-4.2	0.107	0.001	0.666	0.028	0.003	0.115
4.2-6.6	0.059	0.018	0.192	0.027	0.007	0.115
SW-53	7.29	0.777	17.8	1.23	0.093	3.15
Runoff	20.7	0.007	77.7	3.70	0.137	21.8

[a] Concentrations were converted from becquerels per liter to picocuries per liter.

Doses were calculated for an adult, child, and infant. We used the same water ingestion rates for these receptors as defined by the *RAC* scenarios. That is, water ingestion rates of 1, 1.5, and 2 L d^{-1} for 365 days per year were used for the infant, child, and adult rancher scenarios, respectively. We also assumed no filtration was in place so that all suspended plutonium in water was ingested regardless of particle size. ICRP Publication 70 ingestion dose conversion factors were used throughout the calculation.

B.2.2 Results

Measured activity concentrations decreased as a function of depth. Consequently, the deeper the well, the lower the annual dose (Tables B2 through B-4). Doses from the ingestion of seep water were highest and ranged from a minimum of 0.56 mrem to a maximum of 13 mrem for the adult receptor. Doses were higher for adult compared to the child or infant because of higher water ingestion rates. Doses from the ingestion of discharge water ranged from 0.0057 mrem for the infant to 0.0067 mrem for the adult. These doses would be close to those received by offsite individuals who drank water from one of the reservoirs (Standley Lake) that receives water from Woman Creek because it represents mean concentrations in the water discharged from the site. Doses from the ingestion of well water were less than 1 mrem.

There are currently no receptors who consume seep or well water at the site. This calculation is intended to put the potential for radiation dose from groundwater sources into perspective. While we believe the likelihood of using groundwater at the site as a primary drinking water source in the near future is small, doses from drinking such water are near the 15 mrem dose limit. Radionuclides in well and seep water are transient in nature and the measurements represent upper bound values that were measured during a 65-day period while saturated conditions existed in the soils. These conditions do not represent typical conditions at the site and, therefore, these doses must be considered as upper bound estimates, at least in the current time frame. Based on these calculations, additional study and environmental monitoring is recommended to assure radiation dose from the groundwater pathway for future receptors is minimized.

Table B-2. Drinking Water Ingestion Doses from Plutonium

	^{239}Pu–Mean			^{239}Pu–Minimum			^{239}Pu–Maximum		
Well depth or source	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)
2.7-4.2 m	0.061	0.059	0.071	0.00063	0.00061	0.00073	0.38	0.36	0.44
4.2-6.6 m	0.033	0.032	0.039	0.010	0.0097	0.012	0.11	0.11	0.13
SW-53	4.1	4.0	4.8	0.44	0.42	0.51	10	9.7	12
Discharge Water	0.0049	0.0047	0.0058	N/A	N/A	N/A	N/A	N/A	N/A

Table B-3. Drinking Water Ingestion Doses from Americium

Well depth or source	^{241}Am– Mean			^{241}Am– Minimum			^{241}Am–Maximum		
	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)
2.7-4.2 m	0.014	0.012	0.015	0.0013	0.0012	0.0014	0.057	0.051	0.060
4.2-6.6 m	0.014	0.012	0.014	0.0037	0.0033	0.0039	0.057	0.051	0.060
SW-53	0.61	0.55	0.65	0.046	0.041	0.049	1.6	1.4	1.7
Discharge Water	0.00077	0.00069	0.00083	N/A	N/A	N/A	N/A	N/A	N/A

Table B-4. Total Drinking Water Ingestion Doses from Americium and Plutonium

Well depth or source	Total–Mean			Total–Minimum			Total–Maximum		
	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)	Infant (mrem)	Child (mrem)	Adult (mrem)
2.7-4.2	0.075	0.071	0.085	0.0019	0.0018	0.0021	0.43	0.42	0.50
4.2-6.6	0.047	0.045	0.053	0.014	0.013	0.016	0.17	0.16	0.19
SW-53	4.7	4.5	5.4	0.49	0.47	0.56	12	11	13
Discharge Water	0.0057	0.0054	0.0067						

B.4 REFERENCES

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APPENDIX C

EVALUATION OF BACKGROUND CONCENTRATIONS OF PLUTONIUM IN THE ENVIRONMENT AROUND THE ROCKY FLATS PLANT

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EVALUATION OF BACKGROUND CONCENTRATIONS OF PLUTONIUM IN THE ENVIRONMENT AROUND THE ROCKY FLATS PLANT

This appendix originally appeared as Appendix H of the final report on Task 4 of the Rocky Flats Dose Reconstruction project: S.K. Rope, K.R. Meyer, M.J. Case, D.W. Schmidt, T.F. Winsor, and M. Dreicer, *Task 4: Evaluation of Historical Environmental Data*, *RAC* Report #1 CDPHE-RFP-1997-FINAL (1997). The principal contributors to this appendix (in alphabetical order of their last names) are Helen A. Grogan, Susan K. Rope, and Duane W. Schmidt. The appendix is included here with their permission, verbatim, its original formatting and units. Only the headers and footers have been changed to reflect its inclusion in the current report. The reader will find some references in the appendix to other sections in the original report.

APPENDIX C

EVALUATION OF BACKGROUND CONCENTRATIONS OF PLUTONIUM IN THE ENVIRONMENT AROUND THE ROCKY FLATS PLANT

This appendix provides further details about the information presented in Chapter VIII of this report. We describe sources of environmental plutonium around the Rocky Flats Plant (RFP), the isotopic composition of global fallout compared to Rocky Flats plutonium, temporal trends in global fallout, and measured background levels of plutonium in soils around the RFP and across the United States.

A variety of units have been used in the literature for expressing concentrations of plutonium in soils and other media. We have tried to avoid confusion by converting reported concentrations to SI units (becquerels per kilogram [Bq kg^{-1}], becquerels per square meter (Bq m^{-2}], or becquerels per liter [Bq L^{-1}]) when comparing different sets of values. The units used in the original source document are also sometimes provided. In some cases, especially when we are mostly interested in the relative results from a single study, only the units from the original source document are provided. To convert values to different units, consult Table H-17 in the Annex to this appendix.

SOURCES OF BACKGROUND ENVIRONMENTAL PLUTONIUM

Global Fallout from Nuclear Weapons Testing

Atmospheric nuclear weapons testing is the largest source of plutonium in the environment (Harley 1979). Radionuclides formed in nuclear weapons tests are summarized in Holleman et al. (1987). Weapons-grade plutonium is composed primarily (weight-basis) of ^{239}Pu, but it also includes ^{238}Pu, ^{240}Pu, ^{241}Pu, and ^{242}Pu. In nuclear explosions, both fission and fusion weapons release plutonium; these releases are derived from unused plutonium (that does not fission) and from neutron capture reactions, which create the majority of the higher mass isotopes—^{240}Pu, ^{241}Pu, ^{242}Pu, and ^{243}Pu. The quantities of ^{242}Pu and ^{243}Pu produced are very small. Very large quantities of ^{241}Pu are produced. However, ^{241}Pu decays primarily by weak beta emissions, and its radiological impacts are much less significant than those of the primary alpha-emitting plutonium isotopes—^{238}Pu, ^{239}Pu, and ^{240}Pu. Other heavy-element radionuclides are also released in nuclear weapons tests, including ^{237}U, ^{239}Np, and ^{241}Am. Uranium-237 and ^{239}Np have relatively short half-lives of about 7 days and 2 days, respectively, and they would not persist in the environment. Americium-241 builds up in the environment as a result of ^{241}Pu decay; therefore, it is present in the environment in significant quantities relative to the primary alpha-emitting plutonium isotopes. This appendix concentrates on the primary alpha-emitting plutonium isotopes ^{238}Pu, ^{239}Pu, and ^{240}Pu.

Eisenbud (1987) summarizes the history of nuclear weapons testing. Atmospheric testing of nuclear weapons started in New Mexico in July 1945. The majority of tests were performed by the U.S., former Soviet Union, United Kingdom, France, and China. Most of the atmospheric tests were performed in the 1950s and the early 1960s, before the signing of an atmospheric

nuclear weapons test ban agreement in 1963 by the U.S., the former Soviet Union, and the United Kingdom. The announced atmospheric tests are summarized in Table H-1, with estimated yields given in units of megatons of TNT, which would produce an equivalent explosive yield.

Holleman et al. (1987) provides a summary of the atmospheric transport of fallout from nuclear weapons testing. Because nuclear explosions create extremely high temperatures, a fireball is formed after the explosion. The expanding fireball can rise many kilometers, carrying debris from the explosion with it, and it reaches greater altitude for higher yield weapons. At low yields, from about 10 to about 200 kilotons, all of the debris remains in the troposphere (lower atmosphere). At high yields (1 to 2 megatons), 90 to 99% of the debris reaches the stratosphere. Transport of the material is dependent on the height at which the fireball initially injects the debris.

Table H-1. Summary of Announced Atmospheric Nuclear Weapons Tests[a]

Country	Period	Number of tests	Estimated total yield (megatons)
U.S.	1945–1962	193	139
Former Soviet Union	1949–1962	142	358
United Kingdom	1952–1953	21	17
France	1960–1974	45	12
China	1964–1980	22	21
Total		423	547

[a] Adapted from Eisenbud (1987).

Particles formed in the nuclear explosion can be transported long distances by winds. Material reaching the stratosphere is transported around the globe. Deposition of material from the atmosphere occurs by dry or wet deposition. To a great extent, air masses of the northern and southern hemispheres remain separated, but limited exchange between hemispheres does occur.

Other geographic features are relevant to global fallout in the area around the RFP. Mountains alter wind currents, resulting in a downward mixing of higher altitude air, which may increase the ground-level air concentrations of fallout on the lee side of the mountains (the side that is sheltered from the wind). High mountain passes and the lee side of mountains generally receive more precipitation than surrounding areas, which may increase the wet deposition of global fallout. For meteorological reasons, material from the stratosphere is transferred into the troposphere primarily over the middle latitudes (about 40° to 50° latitude) (UNSCEAR 1993). Thus, the greatest amounts of fallout from large tests, which reach the stratosphere, are eventually distributed in the middle latitudes, with lesser amounts distributed toward the poles and the equator (UNSCEAR 1993). The RFP is in the middle latitudes, at a latitude of about 40° north. Distribution of fallout from lower yield tests is dependent on the location of the explosion.

The small particles of debris from atmospheric nuclear weapons tests may remain in the atmosphere for quite some time. For material that reaches the stratosphere, residence times are generally determined to be about 2 to 4 years. Thus, tests that inject debris into the stratosphere generally do not produce the highest ground-level fallout concentrations until about 2 years after the explosions.

Global Fallout from SNAP 9A Burnup

In April 1964, a Transit Navigational Satellite was launched from California. Part of the payload was an auxiliary power generator (called SNAP 9A), which contained 17 kCi (6.3×10^{14} Bq) ^{238}Pu (Harley 1979). The rocket system failed, and the satellite reentered the atmosphere in the southern hemisphere, burning up upon reentry at about 50-km (30-mi) altitude. Essentially all of the plutonium activity was ^{238}Pu. The first arrival in the northern hemisphere of ^{238}Pu fallout from the satellite burnup was measured in early 1966 in Italy. This source of plutonium contributes a small amount to the background total plutonium in the RFP area (see Table H-2 for a general comparison).

Localized Sources

A number of sources of localized plutonium exist in the environment, including both releases from nuclear processing facilities and releases from other accidents (Harley 1979). Releases from localized sources are confined to the lower atmosphere and, thus, are not globally distributed. The Nevada Test Site (NTS), in southern Nevada, was used for test detonations of small nuclear weapons up through 1961; safety tests in which the high explosives in nuclear weapons were detonated (with plutonium in the tested device); Plowshare explosions (using nuclear explosions for peaceful purposes); and accidental venting of underground weapons tests. A considerable amount of unfissioned plutonium was distributed from these tests. Material from the NTS was distributed at least as far away as Salt Lake City, Utah (about 600 km [370 mi] from the NTS), and it may have contributed small amounts to the plutonium deposition in Colorado.

The chemical explosives in nuclear weapons exploded in two incidents, following crashes of U.S. military aircraft. The first occurred in Palomares, Spain, in 1966, and the second in Thule, Greenland, in 1968. Both resulted in local dispersion of plutonium from the weapons.

A number of the U.S. atmospheric nuclear weapons tests were performed in the Pacific Ocean, at Bikini and Enewetak Atolls. Because large quantities of plutonium were produced in these tests, there was significant plutonium deposited in the local area around the tests.

France, India, and the United Kingdom have also released significant quantities of plutonium to oceans, in effluents and as packaged waste for disposal (Harley 1979). Essentially all of this material has remained in the oceans.

Table H-2. Estimated Plutonium Inventories (kCi) in Soils in 1970[a]

Location	Weapons ^{239,240}Pu	Weapons ^{238}Pu	SNAP ^{238}Pu
Northern hemisphere	253 ± 33	6.1 ± 0.8	3.1 ± 0.8
Southern hemisphere	67 ± 14	1.6 ± 0.3	10.3 ± 2.1
Total	320 ± 36	7.7 ± 0.9	13.4 ± 2.2

[a] Based on measurements of plutonium from numerous locations (Harley 1979).

Several U.S. Department of Energy (DOE) weapons plants in the U.S. process plutonium, and releases have occurred from some of them (Harley 1979). At the Mound facility in Ohio, a

liquid release of about 10 Ci (4×10^{11} Bq) of ^{238}Pu occurred in 1969 from a break in a waste pipeline. At the Los Alamos National Laboratory, in New Mexico, about 2 Ci (8×10^{10} Bq) of plutonium has been released to canyon waste disposal sites. Releases of plutonium have also occurred from the Hanford Site in Washington (TSP 1994), the Savannah River Site in South Carolina (RAC 1999), and to a lesser extent from the Oak Ridge National Laboratory in Tennessee (ChemRisk 1993). All of these facilities have been the subject of separate dose reconstruction projects and resulted in more or less localized plutonium deposition.

Table H-2 summarizes the estimated quantities of plutonium in soils in 1970, based on measurements at a number of locations in the northern and southern hemispheres, that were due to weapons tests and the SNAP 9A burnup (Harley 1979). These values are presented to give a rough indication of global inventories of fallout plutonium. In the northern hemisphere, background plutonium concentrations in soils primarily result from $^{239,240}Pu$ in weapons fallout.

ISOTOPIC COMPOSITION OF GLOBAL FALLOUT AND ROCKY FLATS PLUTONIUM

The plutonium processed at the RFP is weapons-grade plutonium, consisting primarily of ^{239}Pu. Plutonium from atmospheric nuclear weapons tests is weapons-grade plutonium that has undergone (partial) fission and neutron capture reactions in the nuclear explosion. Because of these reactions, the relative abundance of the various plutonium isotopes is altered in the exploded material.

Krey and Krajewski (1972) measured the isotopic plutonium composition of a soil sample thought to contain plutonium essentially only from RFP releases and a sample from New York thought to contain only global fallout plutonium. Isotopic ratios, relative to ^{239}Pu, are compared in Table H-3.

Table H-3. Mass Isotopic Ratios, Relative to ^{239}Pu, for Soil Samples Contaminated by Rocky Flats Plutonium or by Global Fallout Plutonium

Sample	$^{238}Pu/^{239}Pu$		$^{240}Pu/^{239}Pu$		$^{241}Pu/^{239}Pu$		$^{242}Pu/^{239}Pu$	
RFP plutonium	7.15 ± 10^{-5}	±9%	5.10 ± 10^{-2}	±0.19%	2.31 ± 10^{-3}	±0.42%	1.43 ± 10^{-4}	±1.7%
Global fallout	1.35 ± 10^{-4}	±5%	1.80 ± 10^{-1}	±0.24%	7.76 ± 10^{-3}	±0.5%	3.89 ± 10^{-3}	±0.59%

[a] Source: Krey and Krajewski (1972).

Other sources have reported slightly different isotopic ratios for global fallout plutonium. Krey (1976) reported measurements for two samples from New York, with an average ratio $^{240}Pu/^{239}Pu$ of 0.163 ± 0.008. Bennett (1978) reported measured plutonium isotopic mass ratios for stratospheric air samples for 1959–1970. The average measured ratio for $^{240}Pu/^{239}Pu$ was 0.18, for $^{241}Pu/^{239}Pu$ during 1963–1967 was 0.0138, for $^{241}Pu/^{239}Pu$ during other years was 0.0118, and for $^{242}Pu/^{239}Pu$ was 0.0034. While there may be slight differences in isotopic ratios in samples of global fallout plutonium, the isotopic ratios for RFP plutonium are significantly different than those for global fallout material. These significant differences can and have been used to differentiate between RFP and global fallout plutonium and to determine which source dominates

in a particular soil sample (see also Table H-10). The most frequently used ratio is $^{240}Pu/^{239}Pu$ because the higher abundance of ^{240}Pu compared to other isotopes results in smaller uncertainties.

TEMPORAL TRENDS IN GLOBAL FALLOUT

In using background concentrations of plutonium in the environment for comparisons with concentrations near the RFP, it can be important to recognize temporal trends (changes with time) in global fallout. The major temporal trend in fallout plutonium concentrations is because of the timing of the weapons tests, which were the fallout plutonium source. Bennett (1978) summarizes the estimated explosive yields of all atmospheric nuclear weapons tests (see Table H-4). The cumulative yield is plotted later in Figure H-2.

Table H-4. Summary of Estimated Total Explosive Yields (Megatons) from Atmospheric Nuclear Tests

Period	Total explosive yield	Cumulative yield
1945–1951	0.75	0.75
1952–1954	60.52	61.27
1955–1956	30.79	92.06
1957–1958	81.39	173.45
1960–1961	122.43	295.88
1962	217.40	513.28
1964–1970	21.23	534.51
1971–1974	6.46	540.97
1976–1978	4.16	545.13

We examine the temporal trends in fallout plutonium by reviewing modeling predictions performed by Bennett (1978). We do not rely on these predictions for explicit, quantitative uses; they are presented to give an appreciation of the general trends.

Bennett (1978) used this information about the timing of weapons testing, the locations of the detonations, and an atmospheric transport model to predict fallout concentrations of plutonium and americium in surface air in the middle latitudes of the northern hemisphere. Table H-5 shows the predicted air concentrations of $^{239,240}Pu$, and Figure H-1 is a plot of these concentrations. The air concentrations of plutonium from global fallout vary considerably over time. It is important to consider this temporal trend of air concentrations when measured concentrations around the RFP are compared to background concentrations. Because of the seasonal changes in mixing of air masses, there are also seasonal trends in fallout air concentrations at ground level (Holleman et al. 1987). For short-term air concentration measurements, these seasonal trends should be considered. Because our major focus of this report is soil samples, the seasonal trends are not examined in more detail.

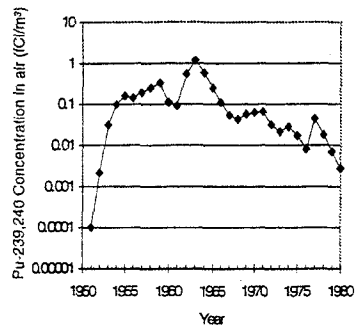

Figure H-1. Predicted surface air concentrations of $^{239,240}Pu$ (fCi m^{-3}) from global fallout from nuclear weapons testing. Predictions are for the mid-latitudes in the northern hemisphere.

Table H-5. Predicted Surface Air Concentrations of $^{239,240}Pu$ (fCi m^{-3}) in Mid-Latitudes of the Northern Hemisphere from Global Fallout from Nuclear Weapons Testing[a]

Year	Concentration	Year	Concentration	Year	Concentration
1950	0	1961	0.089	1972	0.032
1951	0.0001	1962	0.54	1973	0.021
1952	0.0022	1963	1.18	1974	0.028
1953	0.031	1964	0.58	1975	0.017
1954	0.097	1965	0.25	1976	0.0083
1955	0.16	1966	0.11	1977	0.044
1956	0.14	1967	0.054	1978	0.018
1957	0.19	1968	0.042	1979	0.0071
1958	0.25	1969	0.056	1980	0.0028
1959	0.33	1970	0.062		
1960	0.11	1971	0.066		

[a] Source: Bennett (1978).

Bennett (1978) also used the atmospheric transport model to predict deposition rates and cumulative deposition of plutonium and americium in the New York region. These predictions are shown in Table H-6. The predicted cumulative deposition of $^{239,240}Pu$ is plotted in Figure H-2, along with the cumulative yield of the weapons tests for comparison. The predicted cumulative deposition follows the same general shape as the cumulative yield after the lag time because the residence time of the material in the stratosphere is accounted for. The temporal trend in the cumulative deposition of fallout plutonium should be considered when comparing RFP-influenced soil sample results with background results. This trend can be especially important when comparing samples taken at different times. We note that the predicted cumulative deposition of plutonium from nuclear weapons fallout reaches about 90% of its predicted maximum value in 1968, and it reaches 95% of maximum in 1971 (this is relevant to the background soil samples discussed later in this appendix). Although these predicted depositions are for New York, they should also be relevant to the RFP area because both locations are in the middle latitudes.

Table H-6. Predicted Deposition Rate and Cumulative Deposition of $^{239,240}Pu$ in the New York Area because of Global Fallout from Nuclear Weapons Testing[a]

Year	Deposition rate (mCi km^{-2} y^{-1})	Cumulative deposition (mCi km^{-2})	Year	Deposition rate (mCi km^{-2} y^{-1})	Cumulative deposition (mCi km^{-2})
1950	0	0	1968	0.021	2.00
1951	0.00006	0.00039	1969	0.047	2.04
1952	0.0012	0.0013	1970	0.031	2.07
1953	0.017	0.019	1971	0.029	2.10
1954	0.054	0.072	1972	0.023	2.13
1955	0.091	0.16	1973	0.017	2.14
1956	0.077	0.24	1974	0.018	2.16
1957	0.11	0.35	1975	0.012	2.17
1958	0.14	0.49	1976	0.0075	2.18
1959	0.19	0.67	1977	0.024	2.20
1960	0.061	0.73	1978	0.0098	2.21
1961	0.049	0.78	1979	0.0039	2.22
1962	0.30	1.08	1980	0.0016	2.22
1963	0.44	1.52	1981	0.00061	2.22
1964	0.26	1.78	1982	0.00022	2.22
1965	0.11	1.89	1983	0.00011	2.22
1966	0.046	1.93	1984	0.00006	2.22
1967	0.042	1.98			

[a] Source: Bennett (1978).

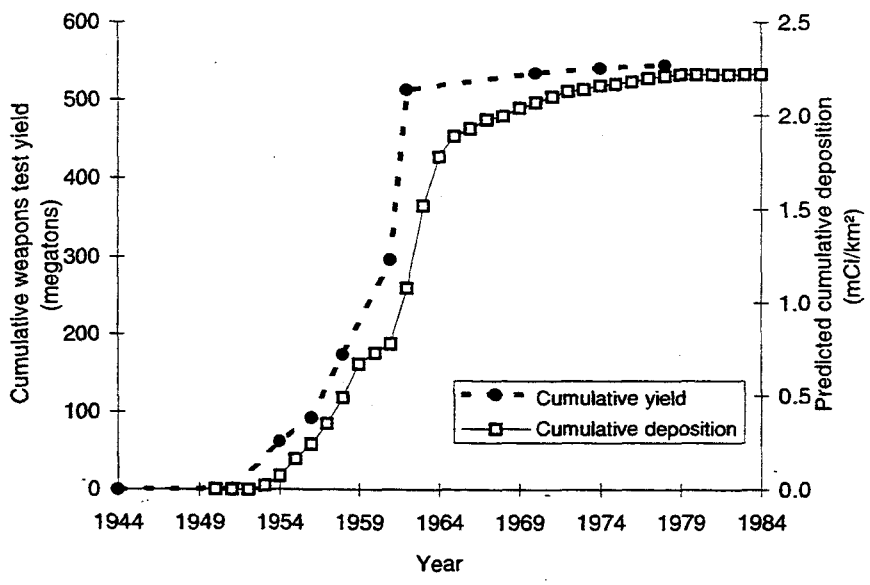

Figure H-2. Predicted cumulative deposition of plutonium in the New York area because of global fallout from nuclear weapons testing. For comparison, the cumulative explosive yield of atmospheric weapons tests is also plotted.

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BACKGROUND PLUTONIUM IN SOIL NEAR THE ROCKY FLATS PLANT

This section describes results from studies around the RFP that represent background concentrations of plutonium in soil. The fire at the RFP in 1969 caused an increased interest in monitoring soil concentrations of plutonium around the plant. A number of soil monitoring studies around the plant were performed or begun in late 1969 and in the early 1970s. Studies were performed by the National Center for Atmospheric Research (NCAR) for the Colorado Committee for Environmental Information (CCEI), the Colorado Department of Health (CDH), the Health and Safety Laboratory (HASL) of the Atomic Energy Commission, and Colorado State University (CSU). In addition, a study by the RFP of background concentrations involves sampling at 50 locations.

The NCAR and CDH obtained surface samples (to 1-cm [0.4-in.] maximum depth) and reported results as mass concentrations, while HASL obtained samples to 10 and 20-cm (4 and 8-in.) depths and reported results as total deposition (per unit area). The shallow depths of the NCAR and CDH samples mean that not all of the plutonium in the soil column was sampled. It is not reasonable to convert the mass concentration results of NCAR and CDH to total deposition values; thus, the NCAR and CDH results cannot be directly compared with HASL results. The CSU study used both surface and deeper soil samples and developed models to describe the soil concentrations of plutonium as a function of depth.

National Center for Atmospheric Research Study

The first study was performed by NCAR for CCEI in late 1969 and early 1970. Results were reported first by CCEI (CCEI 1970), with additional results given in the later report by NCAR (Poet and Martell 1972). This study sampled soils at 35 locations around the RFP and in the Denver area and three locations on the eastern slope of the Rocky Mountains that were thought to contain plutonium only from nuclear weapons fallout. For this study, surface soil samples were taken to a depth of 1 cm (0.4 in.). The background sampling locations are shown in Figure H-3. Results from the background locations are provided in Table H-7. Analysis errors (standard deviations) are included to provide general perspective on the analytical precision. Results were given in units of disintegrations per minute per gram (dpm g^{-1}), and we have converted these to becquerels per kilogram (Bq kg^{-1}) in Table H-7.

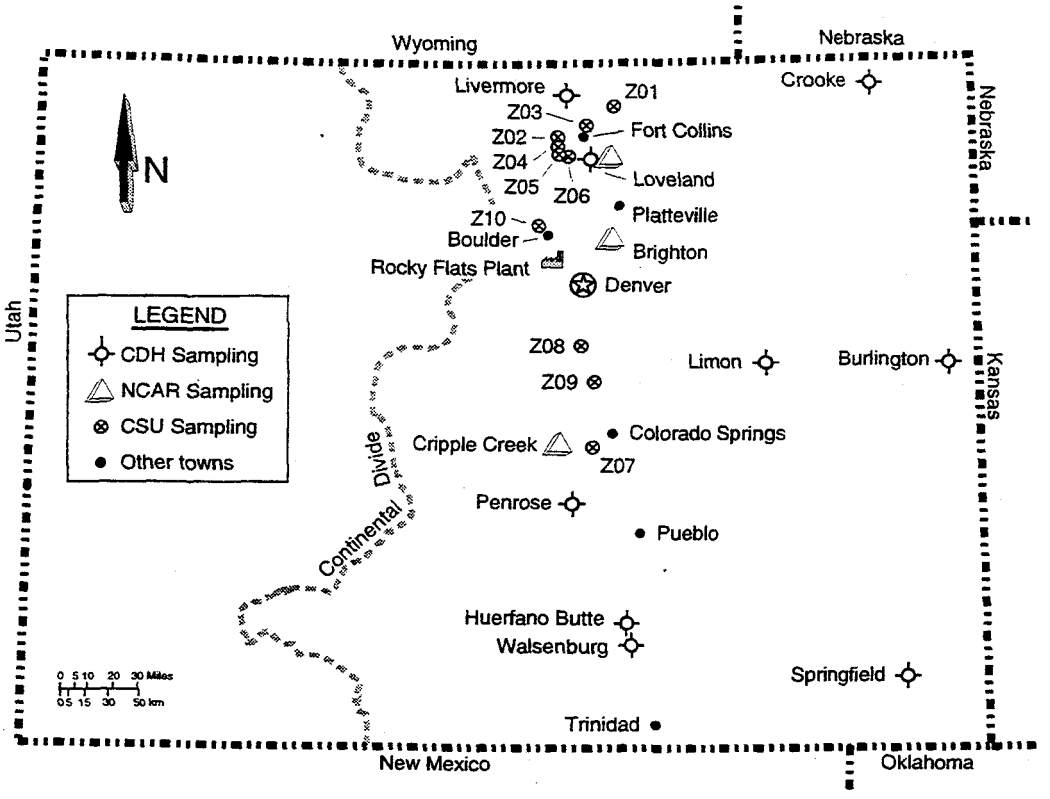

Figure H-3. Background sample locations in Colorado established by the National Center for Atmospheric Research, Colorado Department of Health, and Colorado State University soil studies (after the 1969 fire).

CDH Sampling

Soil sampling around the RFP was also performed by the CDH. Results of monitoring for 1970–1977 are presented by CDH (1977 and 1990) and Jones and Zhang (1994). Samples were generally collected from 13 sectors near the RFP and up to nine remote sites in Colorado each year, although in some years not all the sites were sampled. The remote site locations are shown in Figure H-3. The CDH used its own method to obtain samples for 1975–1988. This method included taking 25 individual surface samples at each site and then compositing to form a single sample for analysis. The sampling procedures used for years before 1975 were not detailed. The sampling depth has changed over the years, with depth 0.16 cm (0.06 in.) used for 1970–1974, 0.32 cm (0.13 in.) for 1975–1981, 0.48 cm (0.19 in.) for 1986, and 0.64 cm (0.25 in.) for 1989 and 1991 (Jones and Zhang 1994). Results from the background locations for 1976 and 1977 are provided in Table H-8. Analysis errors (2 σ) are also shown to provide general perspective on the analytical precision. Results for 1970–1991 are summarized in Table H-9, although no results for these background locations were available for 1974, 1979, 1981–1985, 1987, 1988, and 1990. Results were given in units of disintegrations per minute per gram, which we converted to becquerels per kilogram in Tables H-8 and H-9.

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Table H-7. Background (Fallout) Concentrations of Plutonium in Surface Soil (0–1 cm [0–0.4 in.]) Measured by the National Center for Atmospheric Research around the Rocky Flats Plant in 1969–1970

	Concentration of $^{239,240}Pu$[a]			
	dpm g^{-1}		Bq kg^{-1}	
Location	Value	Std. dev.[b]	Value	Std. dev.[b]
Loveland	0.047	0.013	0.78	0.22
Loveland	0.056	0.025	0.93	0.42
Loveland	0.045	0.008	0.75	0.13
Loveland	0.026	0.006	0.43	0.1
Loveland	0.043	0.005	0.72	0.08
Brighton	0.093	0.009	1.6	0.15
Cripple Creek	0.140	0.027	2.3	0.45
Cripple Creek	0.052	0.012	0.87	0.20
Cripple Creek	0.117	0.015	2.0	0.25
Mean[c]	0.069		1.1	

[a] The source document (Poet and Martell 1972) gives results in units disintegrations per minute per gram (dpm g^{-1}).
[b] Std. dev. = standard deviation.
[c] The arithmetic mean has been calculated, in this present work, from the individual values.

Table H-8. Background (Fallout) Concentrations of $^{239,240}Pu$ in Surface Soil Measured by the Colorado Department of Health in Colorado in 1976 and 1977 (Bq kg^{-1})[a]

	1976		1977	
Location	Value	Counting error (2 Φ)	Value	Counting error (2 Φ)
Burlington	0.3	0.3	1.2	0.3
Crooke	1.3[b]	0.2	0.7	0.3
Limon	0.7	0.3	0.7	0.3
Livermore	0.3	0.3	<0.3	
Loveland	[c]		0.3	0.3
Penrose	1.5	0.7	0.7	0.3
Springfield	0.3	0.3	0.3	0.3
Walsenburg	[c]		0.7	0.7

[a] Values were reported in units disintegrations per minute per gram in the source document (CDH 1977). Sampling depth for these years was 0.32 cm (0.13 in.).
[b] Average of two samples.
[c] No sample was taken at this location in 1976.

Table H-9. Background (Fallout) Concentrations of $^{239,240}Pu$ in Surface Soil Measured by the Colorado Department of Health in Colorado in 1970–1991 (Bq kg^{-1})[a]

Location	1970	1971	1972	1973	1975	1976	1977	1978	1980	1986	1989	1991
Burlington	1.5	1.8	1.2	0.8	[b]	0.3	1.2	<0.7	0.7	<0.3	1.7	0.5
Crooke	0.7	2.2	1.8	0.8	[b]	1.3	0.7	1.2	[b]	<0.3	0.7	0.7
Huerfano Butte	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	[b]	2.3
Limon	2.2	1.0	1.2	1.0	[b]	0.7	0.7	<0.7	0.7	<1.2	0.5	1.0
Livermore	0.7	<0.7	1.2	0.7	[b]	0.3	<0.3	0.7	[b]	<0.7	<0.3	1.3
Loveland	1.8	1.7	[b]	2.0	[b]	[b]	0.3	<0.7	[b]	<0.7	<0.3	<0.3
Penrose	1.8	1.3	1.8	1.0	[b]	1.5	0.7	2.2	[b]	<0.7	0.3	2.5
Springfield	0.7	1.5	2.0	<0.7	0.7	0.3	0.3	4.5	0.7	<1.5	<0.2	1.3
Walsenburg	1.8	1.2	1.8	0.8	0.3	[b]	0.7	<0.7	[b]	<0.7	1.7	0.8

[a] Values were given in units of disintegrations per minute per gram in the source documents (CDH 1977, CDH 1990; Jones and Zhang 1994). Sample depths were 0.16 cm (0.06 in.) for 1970–1974, 0.32 cm (0.13 in.) for 1975–1981, 0.48 cm (0.19 in.) for 1986, and 0.64 cm (0.25 in.) for 1989 and 1991.

[b] No sample result was available for this location for the indicated year.

Health and Safety Laboratory Studies

Studies of plutonium in soil around the RFP by the HASL of the U.S. Atomic Energy Commission were initiated in early 1970. These studies did not separately select background sampling locations as done in the studies described above. Instead, sample locations were chosen at increasing distances from the RFP, and calculation techniques were generally employed to estimate background concentrations. The first study by the HASL is reported by Krey and Hardy (1970). Samples were collected in February 1970 from 33 sites around the RFP, to distances of about 64 km (40 mi), and primarily in easterly directions from the site. Figure H-4 shows the numbered locations (1–33) except for some of those close to the plant. Samples were collected to a depth of 20 cm (8 in.). At some locations, depth profile information was obtained by collecting samples in incremental layers to a total depth of 20 cm (8 in.). Based on limited depth profile information, Krey and Hardy concluded that less than 1% of the total plutonium in soil was deeper than 13 cm (5.1 in.). Results from this study were expressed in units millicuries per square kilometer total deposited plutonium based on the assumption that the measured plutonium (to depth of 20 cm [8 in.]) was the total deposited plutonium. Because the studies described earlier in this section used shallow sample depths, their results cannot be reasonably compared to results of these HASL studies.

Krey and Hardy (1970) did not measure or calculate background plutonium concentrations in soil from their 1970 sampling. They report a background concentration of 1.5 mCi km^{-2} (56 Bq m^{-2}) based on a single measurement in 1965 in Derby, Colorado (Figure H-4).

Seed et al. (1971) performed additional analyses on the data of Krey and Hardy to estimate the background plutonium concentration. Seed et al. plotted the distribution of measured concentrations on log-probability paper. This plot indicated that the distribution appeared to be made up of two separate lognormal distributions: one that represented samples dominated by RFP material and one that represented samples dominated by worldwide fallout plutonium. The data were separated into these two subgroups and replotted. Straight lines (on log-probability plots)

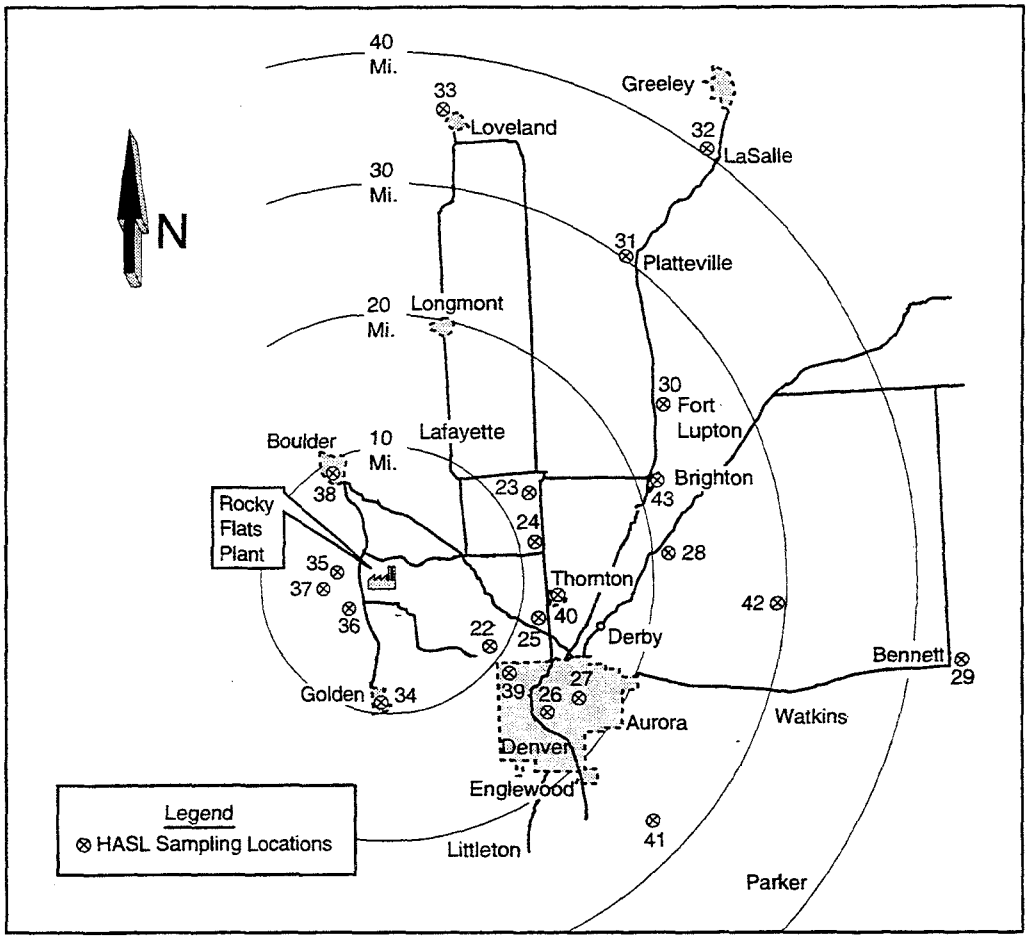

Figure H-4. Locations of the Health and Safety Laboratory soil sampling around the Rocky Flats Plant. Locations 1–33 were used in the 1970 sampling (Krey and Hardy 1970; Seed et al. 1971) and the 1971 sampling (Krey and Krajewski 1972). Locations 34–43 were added in the 1972 sampling (Krey 1976). Only locations numbered higher than 22 are shown here. Other locations are close to the plant.

were fitted to the data to obtain statistics about the distributions. From the fitted lognormal distribution, we determined the background distribution to be represented by a median concentration of 2.3 mCi km^{-2} (85 Bq m^{-2}) and geometric standard deviation 1.16 (Seed et al. [1971] indicated an average value of 2.4 mCi km^{-2} [89 Bq m^{-2}]).

Krey and Krajewski (1972) used isotopic ratios to evaluate RFP and fallout contributions to total plutonium in soil. In October 1971, they obtained additional soil samples from locations 24 and 28 of the previous HASL sampling documented in Krey and Hardy (1970) (see Figure H-4). The new samples were taken to a depth of 10 cm (4 in.). The sample analyses were for isotopic ^{239}Pu and ^{240}Pu, in addition to total ^{239,240}Pu. Ratios of ^{240}Pu to ^{239}Pu were then calculated for the samples at locations 24 and 28, as well as for two "reference" locations known to contain primarily fallout plutonium and primarily RFP plutonium. Because the ratios for RFP plutonium and worldwide fallout plutonium were significantly different, it was possible to calculate the amounts of plutonium that originated from fallout and from the RFP for locations 24 and 28. The total measured ^{239,240}Pu concentrations at locations 24 and 28 were 2.39 (±2.5%) and 1.67

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(±2.5%) mCi km^{-2} (88 and 62 Bq m^{-2}), respectively. For these two locations, the concentrations of ^{239,240}Pu that originated from fallout were then calculated to be 1.49 and 1.52 mCi km^{-2} (55 and 56 Bq m^{-2}). Thus, Krey and Krajewski estimated the background concentration of ^{239,240}Pu because of fallout to be 1.5 mCi km^{-2} (56 Bq m^{-2}). The remaining ^{239,240}Pu in the samples appeared to be due to releases from the RFP.

Krey (1976) applied the isotopic ratio methods of Krey and Krajewski (1972) to an expanded sampling program. In September and October 1972, soil samples were collected from previous locations 22, 23, 27, and 29–32, and from 10 new locations, 34–43 (see Figure H-4). As seen in the figure, these locations ranged from a few kilometers from the RFP to about 64 km (40 mi) from the plant. For this study, sample depth was 10 cm (4 in.), as that depth was thought to contain about 90% of the deposited plutonium. For the analysis, Krey also included the results from locations 24 and 28 from the previous study of Krey and Krajewski (1972). Total measured deposition of ^{239,240}Pu was 1.13–2.87 mCi km^{-2} (41.8–106 Bq m^{-2}). From the ratios of ^{240}Pu to ^{239}Pu, the ^{239,240}Pu deposition from the RFP was calculated. We performed the subtraction to obtain the estimated ^{239,240}Pu deposition from global fallout. The global fallout deposition was 1.12–2.51 mCi km^{-2} (41.4–92.9 Bq m^{-2}). The mean ^{239,240}Pu deposition because of global fallout was calculated by Krey to be 1.7 ±0.5 mCi km^{-2} (63 ±20 Bq m^{-2}). Table H-10 summarizes the estimated background concentrations of plutonium in soils based on the HASL studies.

Table H-10. Summary of Determinations of Background (Global Fallout) Total Deposition of ^{239,240}Pu in Soils within 64 Kilometers (40 Miles) of the Rocky Flats Plant, by the Health and Safety Laboratory

		Deposition of ^{239,240}Pu[a]			
Date	Sites	(mCi km^{-2})	(Bq m^{-2})	Determination method	Reference
1965	1	1.5	56	"Background" location	Krey and Hardy (1970)
1970	33	2.3 ×/÷ 1.16[b]	85 ×/÷ 1.16[b]	Log-probability analysis of distribution of results	Seed et al. (1971)[c]
1971	2	1.50	56	^{240}Pu:^{239}Pu ratios	Krey and Krajewski (1972)
1972	19	1.7 ± 0.5[d]	63 ± 20[d]	^{240}Pu:^{239}Pu ratios	Krey (1976)

[a] Results were reported in source documents in units millicuries per square kilometer.
[b] The ×/÷ value here is one geometric standard deviation of the samples.
[c] Authors were with Dow Chemical. Estimated background was based on their analysis of HASL data.
[d] The ± value here is one standard deviation of the average.

Rocky Flats Plant Routine Sampling

Routine soil sampling for plutonium analyses has been conducted at the RFP from the 1970s. The sampling program has changed over the years, so we examine results from a few select years to determine the usefulness of the data to represent background plutonium concentrations in soil.

In 1972, 20 locations were sampled on each of three concentric rings around the RFP, at 1.6, 3.2, and 8-km (1, 2, and 5-mi) radii (Boss et al. 1973). Surface samples were collected to a depth of 5 cm (2 in.). For the 8-km (5-mi) radius ring, concentrations of ^{239,240}Pu were 5.2–36 Bq kg^{-1}. The uncertainties in these values were extremely large, sometimes greater than 100%.

In 1980, the locations farthest from the center of the site were three locations at the eastern boundary of the site, near Indiana Street (Hornbacher et al. 1981). At each location, nine composite samples were obtained. From the 27 samples at these locations, the concentrations of ^{239,240}Pu were 28–150 Bq kg^{-1}.

In 1991, samples were again taken in concentric rings, although now only at 1.6 and 3.2-km (1 and 2-mi) radii (Altman et al. 1992). Surface samples were collected to a depth of 5 cm (2 in.). For the 3.2-km (2-m) radius ring, concentrations of ^{239,240}Pu were 0.37–130 Bq kg^{-1}. The 1991 annual report (Altman et al. 1992) also summarized results from 1984–1991. For the other years, some had higher maximum concentrations, and one had lower minimum concentrations.

From the results of 1984–1991, some concentrations of ^{239,240}Pu in soil were less than 4 Bq kg^{-1} and, thus, within the range of background seen from other studies. However, none of the sampling locations were specifically chosen to represent background plutonium concentrations unaffected by releases from the RFP. In addition, analyses of the data were insufficient to determine that the measured concentrations were not influenced by plutonium from the RFP. Thus, these data from routine sampling by the RFP may not be as useful as some of the other data in determining the background concentrations. However, the lowest concentrations measured by these studies should provide an indication of background levels.

Colorado State University Study

A CSU study sampled soil extensively from around the RFP during 1992–1994 (Webb 1996). This study included 10 background locations along the front range of the Rocky Mountains, shown in Figure H-3. For the background locations, three different sampling depths were used: 0.3, 3, and 21 cm (0.12, 1.2 and 8 in.). Results of this sampling are given in Table H-11. For the 0 to 21-cm (0 to 8 in.) samples, the result of 3.27 Bq kg^{-1} for location Z10 appeared abnormally high because results for the other locations are 0.22 to 0.62 Bq kg^{-1}. Because of the significant difference in the value for Z10 and the other values, we did not calculate a mean concentration for the 0 to 21-cm (0 to 8-in.) depth. However, there is no information to indicate that the value should be disregarded, and it is probably within the range of statistical variation.

An estimate of the background inventory (total quantity) of plutonium was also described in Webb (1996). Many locations (in addition to the background locations) around the RFP were sampled. At some of these locations, depth profile data were obtained by taking samples in 3-cm (1.2-in.) increments to a depth of 21 cm (8 in.). With concentrations at varying locations and depths, CSU developed models to describe the concentrations as functions of distance and direction from the 903 Area and depth in the soil. These models were then used to develop the following inventory model, which describes the total deposition of plutonium (Webb 1996):

$$I^{\theta,D} = \left(55\ kg\ m^{-2}\right)\left[^{239}Pu\right]^{\theta,D}_{0-3\ cm} \qquad \text{(H-3)}$$

where

$I^{\theta,D}$ = inventory, or total deposition, of ^{239,240}Pu in soil at distance D and direction 2 from the 903 Area (Bq m^{-2})

$\left[^{239}\text{Pu}\right]^{\theta,D}_{0-3\text{ cm}}$ = concentration of plutonium in the 0 to 3-cm layer of soil at distance D and direction 2 from the 903 Area (Bq kg^{-1}).

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Table H-11. Background (Fallout) Concentrations of $^{239,240}Pu$ in Colorado Soils Measured by Colorado State University in 1992–1994 (Bq kg^{-1})[a]

Location	Sample depth		
	0–0.3 cm	0–3 cm	0–21 cm
Z01	0.86	1.20	0.22
Z02	1.52	2.10	0.45
Z03	1.62	1.46	0.33
Z04	1.29	2.10	0.49
Z04[b]	[c]	[c]	0.46
Z05	2.33	2.10	0.51
Z06	0.96	1.14	0.43
Z06[b]	[c]	[c]	0.49
Z07	[c]	3.29	[c]
Z08	1.51	3.22	0.48
Z08[b]	[c]	[c]	0.62
Z09	1.43	2.07	[c]
Z10	2.47	2.70	3.27[d]
Mean	1.55	2.14	[d]
Standard deviation	0.54	0.76	[d]

[a] Source: Webb (1996).
[b] This second result is for a split sample.
[c] No sample.
[d] The value for the 0–21 cm depth at location Z10 seems abnormally high, relative to the other locations; therefore, we did not calculate mean and standard deviation.

Webb (1996) used this equation to calculate the total quantity of background ^{239}Pu in the study area (this was total activity, in giga-becquerel [GBq]). In this present report, we perform essentially the same calculation, but we only calculate the intermediate result of average background $^{239,240}Pu$ deposition (in becquerels per square meter). This is done by applying the equation above to the average background concentration in 0 to 3-cm (0 to .2-in.) soil. The average background concentration in 0 to 3-cm (0 to 1.2-in.) soil is 2.14 Bq kg^{-1}, as used by CSU, and as shown in Table H-11. Thus, the average background deposition is estimated to be (55 kg m^{-2}) × (2.14 Bq kg^{-1}) = 118 Bq m^{-2}. Note that this estimate is based on measured background concentrations for 0 to 3 cm (0.1.2 in.), and on models describing the depth distribution of the plutonium in soil.

EG&G Study

A study by the RFP focused on the characterization of background soils around the RFP (EG&G 1995). This Background Soils Characterization Program (BSCP) included soil sampling at 50 sites remote from the RFP, all in undisturbed areas along the front range of Colorado. Samples were analyzed for concentrations of fallout radionuclides, including $^{239,240}Pu$. In addition,

12 samples were subjected to isotopic analyses so that ratios of ^{240}Pu to ^{239}Pu could be determined. The sampling locations, in relation to the RFP, are shown in Figure H-5.

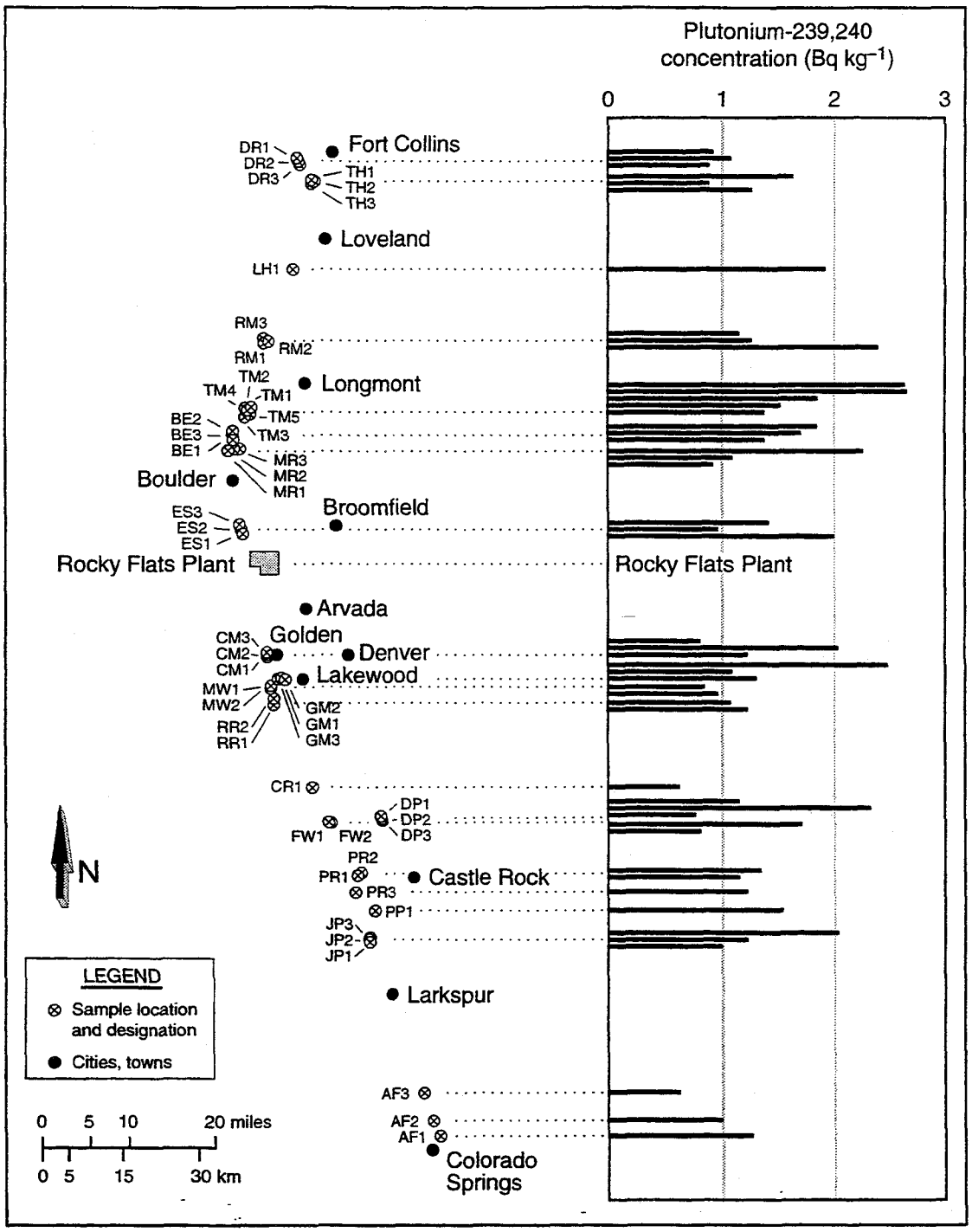

Figure H-5. Background sampling locations and concentrations of $^{239,240}Pu$ in surface soil along the front range, measured in 1994 (EG&G 1995). The left side of this figure shows the sampling locations, to scale. The right side shows the measured plutonium concentrations. For a given location, the measured concentration is shown directly to the right of the location. Data are from Table H-12.

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At each sampling site, two 1 m × 1 m (3.3 × 3.3 ft) square areas were located. From each of the two areas, 5 subsamples were taken, from each of the corners and from the center, and the 10 subsamples were composited to form the sample for analysis. Each of the subsamples was taken to a depth of 5 cm (2 in.), using a 10 cm × 10 cm (4 × 4 in.) square template. Table H-12 shows the results for each sampling location. For some locations, duplicate soil samples were obtained in the field or replicate analyses were performed in the laboratory. In such cases, the values shown in Table H-12 are means of the duplicate or replicate measurements. The result for location GM3 deserves further explanation. The original result for location GM3 appeared to be an outlier. The measured concentration, 0.35 pCi g^{-1}, was 4.6 times higher than the maximum of all other analyses and about 10 times higher than the mean of the other analyses. To investigate the result, two replicate analyses were performed on part of the remainder of the original soil sample, and two replicate analyses were performed on a duplicate field sample. The results of the four additional replicates were 0.025–0.032 pCi g^{-1}, about 10 times lower than the original result. It was concluded that the original result was likely because of a laboratory error rather than to elevated plutonium at the sampled location (EG&G 1995). Thus, the original, high result is not included in the mean shown in Table H-12.

Table H-12. Background Concentrations of $^{239,240}Pu$ in Surface Soil Along the Front Range, Measured by EG&G in 1994 (Bq kg^{-1})[a]

Site	Concentration	Site	Concentration	Site	Concentration	Site	Concentration
AF1	1.26	DR1	0.92	JP3	2.04	RM3	1.15[b]
AF2	1.00	DR2	1.07	LH1	1.92	RR1	1.22
AF3	0.63	DR3	0.89	MR1	0.92	RR2	1.07
BE1	1.37	ES1	2.00	MR2	2.26[b]	TH1	1.63
BE2	1.85	ES2	0.96	MR3	1.09[b]	TH2	0.89
BE3	1.70	ES3	1.41	MW1	0.85	TH3	1.26
CM1	1.22	FW1	0.81	MW2	0.96	TM1	2.65[c]
CM2	2.04	FW2	1.70	PP1	1.54[b]	TM2	1.85
CM3	0.81	GM1	2.48	PR1	1.15	TM3	1.37
CR1	0.63[c]	GM2	1.30	PR2	1.33	TM4	2.66
DP1	1.15	GM3	1.08[d]	PR3	1.22	TM5	1.52
DP2	2.33	JP1	1.00	RM1	2.40		
DP3	0.78	JP2	1.22[c]	RM2	1.26[b]		

[a] Measurements have been converted from units of picocuries per gram , in the original reference (EG&G 1995), to becquerels per kilogram.
[b] Mean of results of two replicate analyses.
[c] Mean of results of duplicate soil samples.
[d] Mean of results of two replicate analyses of each of two duplicate soil samples. Original result, which was determined to be erroneous, is not included (see text for details).

The results of the background concentrations from the BSCP are also shown in Figure H-5, which depicts the plutonium concentrations in relation to the sampling locations. From the 50 sampling locations, the concentrations of $^{239,240}Pu$ ranged from 0.629 to 2.664 Bq kg^{-1}. The mean concentration was 1.40 Bq kg^{-1}, and the standard deviation was 0.54 Bq kg^{-1}. One important conclusion from Figure H-5 is that there does not appear to be any significant trend in plutonium concentration with distance from the RFP, which indicates that the sampling locations probably

represent background concentrations not affected by releases from the RFP. From the 12 samples analyzed isotopically, the mean $^{240}Pu/^{239}Pu$ ratio was 0.1552, and the standard deviation was 0.0093 (EG&G 1995).

Summary of Background Plutonium in Soil near the Rocky Flats Plant

In summary, the measurements performed by NCAR, CDH, CSU, and EG&G (Tables H-7, H-8, H-9, H-11, and H-12) indicate that the background mass concentration of $^{239,240}Pu$ in surface soils (0.16 to 5-cm [0.06 to 2-in.] depths) of eastern Colorado is in the range of about 0.3–4.5 Bq kg^{-1} (0.008–0.1 pCi g^{-1}), although only one value was greater than 3.3 Bq kg^{-1}. The wide variability in results may be due to the very shallow surface layers of soil that were sampled and the spatial patterns of fallout deposition across the large area covered by sampling. To summarize the deposition measurements and calculations performed by HASL (and the analysis of HASL results by Seed et al. [1971]) and by CSU we used the values from Tables H-10 and H-11. For the HASL values, we also included one standard deviation or one geometric standard deviation (where available) to represent likely ranges (Table H-10). These results indicate that the total deposition of $^{239,240}Pu$ from global fallout, in the general area around the RFP and along the front range, was probably in the range of 40–120 Bq m^{-2} (1.1–3.2 mCi km^{-2}).

BACKGROUND PLUTONIUM IN SOIL AT GREATER DISTANCES FROM THE ROCKY FLATS PLANT

This section describes measurements of background soil concentrations of plutonium for locations farther from the RFP. While concentrations at great distances from the RFP may not be indicative of the background around the plant, they do provide some perspective as to how local background concentrations compare with regional and global background.

Purtymon et al. (1990) reports data on soil concentrations of plutonium in northern New Mexico and southern Colorado, which are in the same general region as the RFP, and within 5° latitude. In this study, six locations were sampled in 1981 and 1983, and nine separate locations were sampled in 1986. The locations were all east of (or on) the continental divide. The northern-most location was Monarch Pass, Colorado, about 160 km (100 mi) from the RFP, and the southern-most location was Santa Ana Pueblo, New Mexico, about 480 km (300 mi) from the RFP. Some of the locations are, however, within about 32 km (20 mi) of the Los Alamos National Laboratory, which is a potential source of plutonium in the environment. The soil samples were composites made up of five subsamples, taken to a depth of 5 cm (2 in.). Alpha spectroscopy measurements were performed to obtain ^{238}Pu and $^{239,240}Pu$, which were summed to obtain total plutonium. On the average, ^{238}Pu contributed less than 5% to the total plutonium activity measured. We only consider the $^{239,240}Pu$ measurements here for comparability with other measurements. Concentrations of $^{239,240}Pu$ ranged from 1.2 to 81 fCi g^{-1} (0.044 to 3.0 Bq kg^{-1}), with an average of 14 fCi g^{-1} (0.53 Bq kg^{-1}) and a standard deviation of 18 fCi g^{-1} (0.66 Bq kg^{-1}). The two highest values occurred for locations in high mountain passes on the continental divide. As discussed earlier, higher values are expected for high mountain areas, and it may be reasonable to consider these locations grouped separately from the remaining locations. If these highest values are disregarded, the remainder cover the range of 1.2–19 fCi g^{-1} (0.044–0.71

Bq kg^{-1}), with an average of 9.4 fCi g^{-1} (0.35 Bq kg^{-1}) and a standard deviation of 5.4 fCi g^{-1} (0.20 Bq kg^{-1}).

As mentioned above, Holleman et al. (1987) provides an extensive compilation of datasets on worldwide fallout of plutonium from weapons tests. From this compilation, we extracted measured concentrations of $^{239,240}Pu$ in soil in the U.S. Values were given in units of total deposition (becquerels per square meter) and mass concentration (becquerels per kilogram). Information about individual measurements is given in Table H-16, at the end of this appendix. Holleman et al. does not provide information about sample depths, but this is not necessary for our work. The time the samples were collected is not that important because the earliest date (1962) is after the majority of plutonium was deposited (Figure H-2). The results are summarized by state in Table H-13; for some states only one measurement was available.

Table H-13. Summary of $^{239,240}Pu$ Deposition and Mass Concentrations in Soil in the United States (from the Compilation by Holleman et al. 1987)

				Deposition (Bq m^{-2})[a]			Concentration (Bq kg^{-1})[a]		
State	Sites	Samples	Dates	Min	Max	Avg	Min	Max	Avg
Alaska	5	9	1964–1976	1.18	34	13			
California	3	5	1970–1972	27	37	32			
Colorado	1	7	1965–1970	2.11	67	47			
Florida	1	1	1970			37			
Hawaii	1	1	1970			148			
Illinois	17	62	1970–1981	10.36	256.78	51			
Kansas	1	1	1970			89			
Maine	1	1	1970			63			
Massachusetts	1	1	1972			85			
Michigan	1	1	1976			99.9			
Montana	1	1	1965			70			
New Mexico	6	36	1974–1977				0.0	0.78	0.28
New York	7	16	1964–1973	67	99.9	84			
North Carolina	1	1	1970			89			
Ohio	1	25	1974				0.114	1.528	0.28
Oklahoma	1	1	1970			81			
South Dakota	2	2	1965–1970	85	93	89			
Texas	2	2	1970	32.6	36.6	35			
Utah	1	1	1970			96			
Washington	2	7	1970–1971	1.5	52	20			
Wisconsin	1	1	1972			58			

[a] Min = minimum, max = maximum, and avg = arithmetic average. Minimum and maximum values are taken from the source document (Holleman et al. 1987), and the averages are calculated by us, in this present work. The values presented retain the number of significant figures used by Holleman et al., though we acknowledge that in some cases they are excessive.

The samples in Ohio were taken near the Mound facility, which processed plutonium; however, this facility handled primarily ^{238}Pu. The very low values of deposition reported for Alaska are probably because of the more northerly latitude of Alaska. The single result for Hawaii is relatively high compared to other states closest in latitude (Florida and Texas). The elevated value may be due to Hawaii's proximity to some of the weapons tests in the Pacific and, thus, may reflect some regional (in addition to global) fallout. The minimum deposition value for Colorado (2.11 Bq m^{-2}) does not appear to be a credible value; deposition this small seems

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extremely unlikely. However, the average value for Colorado is in reasonable agreement with other states at similar latitudes (e.g., Illinois).

Because global fallout deposition is correlated with latitude in the northern hemisphere, we also summarized the $^{239,240}Pu$ measurements by latitude bands. We centered the bands around latitude 40 °N because that is the approximate location of the RFP. Table H-14 shows this summary.

CONCLUSIONS ON BACKGROUND PLUTONIUM IN SOILS

From the studies presented here, the measured levels of plutonium in soil around the RFP and around the U.S. are compared in Table H-16. (We acknowledge that this is not necessarily a complete compilation of such data.) Figure H-6 compares the total plutonium deposition for background locations around the RFP to locations in the U.S. at similar latitudes. The range of measured deposition of $^{239,240}Pu$ around the RFP (40–120 Bq m^{-2}, or 1.1–3.2 mCi km^{-2}) is within that seen for other states in the 37.5–42.5 °N latitude range (10–260 Bq m^{-2}, or 0.27–7 mCi km^{-2}), although they tend slightly toward the higher end of measured concentrations. The measured mass concentrations of $^{239,240}Pu$ around the RFP (<0.3–4.5 Bq kg^{-1}, or 0.008–0.1 pCi g^{-1}) exceed (slightly) the ranges of values seen in New Mexico and Ohio measurements (0–3.0 Bq kg^{-1}, or 0–0.08 pCi g^{-1}). Many of the lowest values for the U.S. are for locations, such as Alaska, not in the middle latitudes. Thus, it appears that while measured background concentrations of plutonium in soil around the RFP tend to be higher than background concentrations for many locations, they are still within the ranges observed in other states at similar latitudes.

Table H-14. Summary of $^{239,240}Pu$ Deposition (Bq m^{-2}) and Mass Concentrations (Bq kg^{-1}) in Soil in the United States, by Latitude (from the Compilation in Holleman et al. 1987)

	Deposition[a]					Concentration[a]				
Latitude band	Sites	Samples	Min	Max	Avg	Sites	Samples	Min	Max	Avg
<32.5 °N [b]	3	3	33	37	35					
32.5–37.5 °N	5	7	27	89	47	6	36	0	0.78	0.28
37.5–42.5 °N[c]	30	91	10	260	60	1	25	0.11	1.5	0.28
42.5–47.5 °N	5	10	1.5	93	37					
>47.5 °N[d]	5	9	1.2	34	13					

[a] Min = minimum, max = maximum, and avg = arithmetic average. Minimum and maximum values are taken from the source document (Holleman et al. 1987), and the averages are calculated by us, in this present work.
[b] The single result for Hawaii is not included in this summary, because the value is probably not representative of *global* fallout.
[c] The extremely low value from Colorado is not included in this summary.
[d] All values in this latitude band are from Alaska, at latitudes greater than 60 °N.

Some important characteristics related to plutonium in soils should be considered in evaluations of soil sample results around the RFP. The measurement technique, and more specifically the plutonium isotopes actually measured, should be determined. Essentially all measurements of "^{239}Pu" are actually measurements of $^{239,240}Pu$ because alpha spectroscopy is commonly used for the analyses. If isotopic results, such as the ratio $^{240}Pu/^{239}Pu$, are available, it may be feasible to determine more accurately whether the source of the plutonium is truly background or if it has been influenced by RFP sources. When comparing samples near the RFP to background samples, the time of sample collection can be important because there are temporal trends in the global fallout of plutonium from nuclear weapons testing. Finally, depth distributions of plutonium should be considered relative to the goals of a particular sampling program or analysis. Soil samples taken from the surface soils (e.g., to 1 cm [0.4 in.] or so) are generally not representative of the total deposition of plutonium that exists in the soil column. Quantitative comparisons between results of sampling programs with widely disparate sample depths should be performed when information is available to develop relationships between soil layers of different depths.

Table H-15. Comparison of Measured Deposition and Mass Concentrations of $^{239,240}Pu$ Around the Rocky Flats Plant with those Around the United States

	Deposition (Bq m^{-2})		Concentration (Bq kg^{-1})	
Locations	Minimum	Maximum	Minimum	Maximum
Around RFP	40	120	<0.3	4.5
37.5–42.5 °N Latitude[a]	10	260	0.11	1.5
U.S.	1.2	260	0.0[b]	3.0[b]

[a] The extremely low value from Colorado is not included in this summary.
[b] Mass concentrations were from two states only: New Mexico and Ohio.

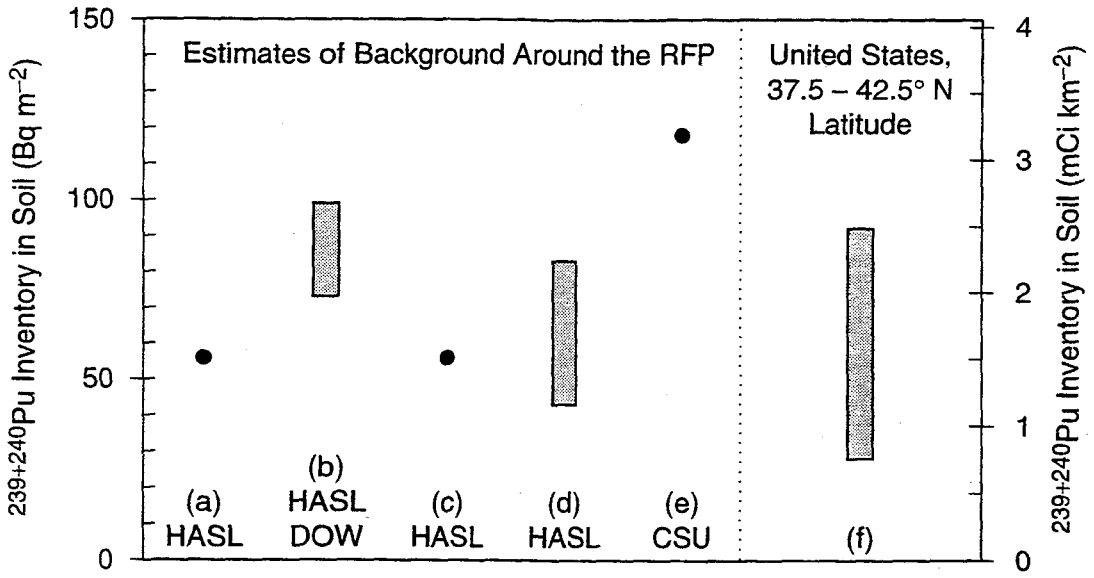

Figure H-6. Background $^{239,240}Pu$ total deposition (inventory) in soils: comparison of levels around the RFP with levels in the U.S. at similar latitude. Data are described in this appendix. Notes: (a) single location, in 1965, (b) examined distribution of measurements from 1970, (c) used $^{240}Pu/^{239}Pu$ ratios for samples from 1971, (d) used $^{240}Pu/^{239}Pu$ ratios for samples from 1972, (e) samples from 1992–1994, 0–3-cm (0–1.2-in.) depth, with depth distribution model, (f) from compilation of numerous measurements.

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ANNEX

Table H-16. Individual Measurements of $^{239,240}Pu$ in Soil in the United States (from the Compilation of Holleman et al. 1987)[a]

State	Location	Latitude (°N)	Date	$^{239,240}Pu$ concentration Bq m^{-2}	Bq kg^{-1}
Alaska	Anaktuvuk Pass	68.10	Jul 1975	5.62	
Alaska	Anaktuvuk Pass	68.10	Jul 1976	1.55	
Alaska	Anaktuvuk Pass	68.10	Sep 1976	1.18	
Alaska	Barrow	71.17	Aug 1964	12.20	
Alaska	Barrow	71.17	1970	14.8	
Alaska	Bettles	66.55	Jul 1976	4.26	
Alaska	Fairbanks	64.51	Jul 1976	8.21	
Alaska	Fairbanks	64.51	1970	31.4	
Alaska	Palmer	61.36	1970	34	
California	Burbank	34.2 [b]	1970	27	
California	Oakland	37.47	Oct 1972	30.00	
California	Oakland	37.47	Oct 1972	30.00	
California	San Francisco	37.48	Oct 1972	34.00	
California	San Francisco	37.48	Oct 1972	37.00	
Colorado	Denver	39.43	Sep 1965	56	
Colorado	Denver	39.43	Feb 1970	32.9	
Colorado	Denver	39.43	Feb 1970	40.7	
Colorado	Denver	39.43	Sep 1970	65	
Colorado	Denver	39.43	Sep 1970	2.11	
Colorado	Denver	39.43	Oct 1970	67	
Colorado	Denver	39.43	1970	67	
Florida	Ft. Pierce	27.27	1970	37	
Hawaii	Papaikou	19.47	1970	148	
Illinois	Argonne	41.43	1970	78	
Illinois	Brookfield	41.49	Sep 1972	57.35	
Illinois	Brookfield	41.49	Oct 1974	65.86	
Illinois	Brookfield	41.49	Jun 1976	70.3	
Illinois	Brookfield	41.49	Jun 1979	36.63	
Illinois	Brookfield	41.49	Oct 1980	49.21	
Illinois	Channahon	41.26	Jun 1978	49.6	
Illinois	Channahon	41.26	Jun 1979	31.08	
Illinois	Channahon	41.26	Jun 1980	19.61	
Illinois	Channahon	41.26	Jun 1981	25.2	
Illinois	Downers Grove	41.48	Jun 1979	18.5	
Illinois	Downers Grove	41.48	Oct 1981	29.2	
Illinois	Dresden Lock and Dam	41.20	Oct 1976	74	
Illinois	Dresden Lock and Dam	41.20	Oct 1978	45.1	
Illinois	Dresden Lock and Dam	41.20	Oct 1979	15.91	
Illinois	Dresden Lock and Dam	41.20	Oct 1980	10.36	
Illinois	Dresden Lock and Dam	41.20	Oct 1981	41.8	
Illinois	Hinsdale	41.48	Oct 1974	127.65	
Illinois	Hinsdale	41.48	Jun 1976	81.4	
Illinois	Lemont	41.40	Nov 1974	56.61	
Illinois	Lemont	41.40	Oct 1978	21.5	
Illinois	Lemont	41.40	Jun 1980	19.61	
Illinois	Lemont	41.40	Oct 1981	23.7	

Table H-16. Individual Measurements of $^{239,240}Pu$ in Soil in the United States, from the Compilation of Holleman et al. (1987) (continued)

State	Location	Latitude (°N)	Date	$^{239,240}Pu$ concentration Bq m^{-2}	Bq kg^{-1}
Illinois	McGinnis Slough	41.39	Sep 1972	72.52	
Illinois	McGinnis Slough	41.39	May 1974	80.3	
Illinois	McGinnis Slough	41.39	Oct 1978	42.5	
Illinois	McGinnis Slough	41.39	Jun 1980	22.57	
Illinois	Mckinley Woods State Park	41.45	Jun 1972	40.7	
Illinois	Mckinley Woods State Park	41.45	Oct 1974	77.7	
Illinois	Mckinley Woods State Park	41.45	Jun 1976	114.7	
Illinois	Mckinley Woods State Park	41.45	Oct 1978	54.8	
Illinois	Mckinley Woods State Park	41.45	Oct 1979	35.52	
Illinois	Mckinley Woods State Park	41.45	Oct 1980	20	
Illinois	Mckinley Woods State Park	41.45	Oct 1981	69.2	
Illinois	Morris	41.22	May 1974	75.85	
Illinois	Morris	41.22	May 1974	256.78	
Illinois	Morris	41.22	Jun 1978	52.2	
Illinois	Morris	41.22	Jun 1979	32.56	
Illinois	Morris	41.22	Jun 1980	27	
Illinois	Morris	41.22	Jun 1981	17	
Illinois	Naperville	41.47	Jun 1972	55.5	
Illinois	Naperville	41.47	May 1974	94	
Illinois	Naperville	41.47	Jun 1978	57.7	
Illinois	Naperville	41.47	Jun 1981	24	
Illinois	Romeoville	41.39	Oct 1978	57.3	
Illinois	Romeoville	41.39	Oct 1981	44.4	
Illinois	Saganashkee Slough	41.41	Jun 1972	77.33	
Illinois	Saganashkee Slough	41.41	May 1974	72.52	
Illinois	Saganashkee Slough	41.41	Oct 1978	27	
Illinois	Saganashkee Slough	41.41	Jun 1980	21.83	
Illinois	Starved Rock State Park	41.19	May 1974	76.22	
Illinois	Starved Rock State Park	41.19	Jun 1978	43.3	
Illinois	Starved Rock State Park	41.19	Jun 1979	31.08	
Illinois	Starved Rock State Park	41.19	Jun 1980	17.39	
Illinois	Starved Rock State Park	41.19	Jun 1981	43.7	
Illinois	Western Springs	41.47	Jun 1979	35.9	
Illinois	Western Springs	41.47	Oct 1980	24.8	
Illinois	Willow Springs	41.50	Oct 1976	107.3	
Illinois	Willow Springs	41.50	Jun 1978	27	
Illinois	Willow Springs	41.50	Oct 1979	30.71	
Illinois	Woodridge	41.46	Oct 1979	32.56	
Illinois	Woodridge	41.46	Jun 1981	30.3	
Kansas	Manhattan	39.11	1970	89	
Maine	Orono	44.53	1970	63	
Massachusetts	North Eastham, Cape Cod	41.52	Oct 1972	85	
Michigan	St. Joseph	42.06	Oct 1976	99.9	
Montana	Bozeman	45.41	Sep 1965	70	
New Mexico	Bernalillo	35.30	Jul 1974		0.22
New Mexico	Bernalillo	35.30	May 1975		0.44
New Mexico	Bernalillo	35.30	Oct 1975		0.04
New Mexico	Bernalillo	35.30	Apr 1976		0.15
New Mexico	Bernalillo	35.30	Oct 1976		0.07
New Mexico	Bernalillo	35.30	Mar 1977		0
New Mexico	Bernalillo	35.30	Oct 1977		0.07
New Mexico	Chamita	36.00	Jul 1974		0.22

Table H-16. Individual Measurements of $^{239,240}Pu$ in Soil in the United States, from the Compilation of Holleman et al. (1987) (continued)

State	Location	Latitude (°N)	Date	$^{239,240}Pu$ concentration Bq m^{-2}	Bq kg^{-1}
New Mexico	Chamita	36.00	Oct 1975		0.63
New Mexico	Chamita	36.00	Mar 1976		0.3
New Mexico	Chamita	36.00	Oct 1976		0.52
New Mexico	Chamita	36.00	Mar 1977		0.63
New Mexico	Chamita	36.00	Oct 1977		0.37
New Mexico	Cochiti	35.45	May 1975		0.07
New Mexico	Cochiti	35.45	Oct 1975		0
New Mexico	Cochiti	35.45	Apr 1976		0.15
New Mexico	Cochiti	35.45	Oct 1976		0.11
New Mexico	Cochiti	35.45	Mar 1977		0.04
New Mexico	Cochiti	35.45	Oct 1977		0.11
New Mexico	Embudo	36.00	Jul 1974		0.19
New Mexico	Embudo	36.00	May 1975		0.3
New Mexico	Embudo	36.00	Oct 1975		0.33
New Mexico	Embudo	36.00	Mar 1976		0.44
New Mexico	Embudo	36.00	Oct 1976		0.7
New Mexico	Embudo	36.00	Mar 1977		0.4
New Mexico	Embudo	36.00	Oct 1977		0.56
New Mexico	Jemez	35.45	Jul 1974		0.04
New Mexico	Jemez	35.45	May 1975		0.04
New Mexico	Jemez	35.45	Sep 1975		0.44
New Mexico	Jemez	35.45	Apr 1976		0.07
New Mexico	Jemez	35.45	Oct 1976		0.26
New Mexico	Jemez	35.45	Mar 1977		0.7
New Mexico	Jemez	35.45	Oct 1977		0.04
New Mexico	Otowi	35.50	Jul 1974		0.44
New Mexico	Otowi	35.50	May 1975		0.22
New Mexico	Otowi	35.50	Oct 1977		0.78
New York	Bronx	40.49	Jul 1970	92.5	
New York	Bronx	40.49	Aug 1970	81.4	
New York	Brookhaven National Laboratory	40.54	Sep 1970	96	
New York	Brookhaven National Laboratory	40.54	Sep 1970	78	
New York	Brookhaven National Laboratory	40.54	Nov 1972	99.9	
New York	Brookhaven National Laboratory	40.54	Nov 1972	91.8	
New York	Brookhaven National Laboratory	40.54	Nov 1972	90.6	
New York	Brookhaven National Laboratory	40.54	Nov 1972	81	
New York	Brookhaven National Laboratory	40.54	1972	88.8	
New York	Brooklyn	40.42	Nov 1972	78	
New York	Fordham University	40.51	Dec 1969	74	
New York	Fordham University	40.51	Jan 1970	81.4	
New York	Fordham University	40.51	Jan 1970	96	
New York	Kitchawan, Westchester County	41.15	Jun 1973	70.3	
New York	New York	40.43	Dec 1964	67	
New York	New York	40.43	1970	96	
New York	Teatown, Westchester County	41.15	Jun 1973	70.3	
North Carolina	Raleigh	35.47	1970	89	
Ohio	Miamisburg	39.38	Oct 1974		0.177
Ohio	Miamisburg	39.38	Oct 1974		0.222
Ohio	Miamisburg	39.38	Oct 1974		0.166
Ohio	Miamisburg	39.38	Oct 1974		0.269
Ohio	Miamisburg	39.38	Oct 1974		0.206
Ohio	Miamisburg	39.38	Oct 1974		0.171

Table H-16. Individual Measurements of $^{239,240}Pu$ in Soil in the United States, from the Compilation of Holleman et al. (1987) (continued)

State	Location	Latitude (°N)	Date	$^{239,240}Pu$ concentration	
				Bq m^{-2}	Bq kg^{-1}
Ohio	Miamisburg	39.38	Oct 1974		0.256
Ohio	Miamisburg	39.38	Oct 1974		0.171
Ohio	Miamisburg	39.38	Oct 1974		0.2
Ohio	Miamisburg	39.38	Oct 1974		0.129
Ohio	Miamisburg	39.38	Oct 1974		0.17
Ohio	Miamisburg	39.38	Oct 1974		0.135
Ohio	Miamisburg	39.38	Oct 1974		0.207
Ohio	Miamisburg	39.38	Oct 1974		0.114
Ohio	Miamisburg	39.38	Oct 1974		0.174
Ohio	Miamisburg	39.38	Oct 1974		0.191
Ohio	Miamisburg	39.38	Oct 1974		0.179
Ohio	Miamisburg	39.38	Oct 1974		0.18
Ohio	Miamisburg	39.38	Oct 1974		0.19
Ohio	Miamisburg	39.38	Oct 1974		0.14
Ohio	Miamisburg	39.38	Oct 1974		0.213
Ohio	Miamisburg	39.38	Oct 1974		0.208
Ohio	Miamisburg	39.38	Oct 1974		0.16
Ohio	Miamisburg	39.38	Oct 1974		1.214
Ohio	Miamisburg	39.38	Oct 1974		1.528
Oklahoma	Tulsa	36.09	1970	81	
South Dakota	Rapid City	44.05	Sep 1965	93	
South Dakota	Vermillion	42.47	1970	85	
Texas	Kingsville	27.31	1970	36.6	
Texas	Weslaco	26.09	1970	32.6	
Utah	Salt Lake City	40.46	1970	96	
Washington	Hanford Reservation	46.50[b]	Feb 1971	19.20	
Washington	Hanford Reservation	46.50[b]	Feb 1971	28.10	
Washington	Hanford Reservation	46.50[b]	Feb 1971	24.00	
Washington	Hanford Reservation	46.50[b]	Feb 1971	8.1	
Washington	Hanford Reservation	46.50[b]	Feb 1971	1.5	
Washington	Hanford Reservation	46.50[b]	Feb 1971	7.8	
Washington	Puyallup	47.11	1970	52	
Wisconsin	Lake Delavan	42.38	Oct 1972	58.46	

[a] The values presented here retain the number of significant figures used by Holleman et al. (1987).
[b] The latitude given in Holleman et al. (1987) appeared to be an error. We have estimated the latitude from maps, and show the estimated latitude here.

Table H-17. Units Conversion Factors

Multiply value with units of:	by:	To obtain value with units of:
	Units of activity	
dpm[a]	0.0167	Bq
dpm	0.45	pCi
pCi	0.037	Bq
pCi	2.22	dpm
pCi	1000	fCi
fCi	3.7×10^{-5}	Bq
fCi	0.001	pCi
Ci	37	GBq
Bq	60	dpm
Bq	27	pCi
Bq	27,000	fCi
	Concentration units: activity per mass	
dpm g^{-1}	16.7	Bq kg^{-1}
dpm g^{-1}	0.45	pCi g^{-1}
pCi g^{-1}	37	Bq kg^{-1}
pCi g^{-1}	2.22	dpm g^{-1}
fCi g^{-1}	0.037	Bq kg^{-1}
Bq kg^{-1}	0.060	dpm g^{-1}
Bq kg^{-1}	0.027	pCi g^{-1}
Bq kg^{-1}	27	fCi g^{-1}
	Concentration units: activity per area	
mCi km^{-2}	37	Bq m^{-2}
Bq m^{-2}	0.027	mCi km^{-2}
	Concentration units: activity per liquid volume	
pCi L^{-1}	0.037	Bq L^{-1}
pCi L^{-1}	1000	fCi L^{-1}
fCi L^{-1}	0.001	pCi L^{-1}
dpm L^{-1}	0.0167	Bq L^{-1}
dpm L^{-1}	450	fCi L^{-1}
Bq L^{-1}	27	pCi L^{-1}
Bq L^{-1}	27,000	fCi L^{-1}

[a] dpm = disintegrations per minute.

Example: The following is an example of using this table of units conversion factors. If you have a value of 120 Bq kg^{-1} and wish to convert to units of pCi g^{-1}, look in the first column to find the units you have (Bq kg^{-1}). Look in the third column to find the row that contains the units to which you want to convert (pCi g^{-1}). Use the conversion factor in the second column that corresponds with that row. In this example, that row of the table shows a factor of 0.027. Multiply the starting value (120) by 0.027 ($120 \times 0.027 = 3.24$). Thus, our starting value of 120 Bq kg^{-1} is equal to 3.2 pCi g^{-1}, with rounding.

APPENDIX D

THE EFFECT OF TIME ON THE RSAL

APPENDIX D

THE EFFECTS OF TIME ON THE RSAL

This time at which the maximum doses occur was examined to ensure that the limiting scenario had been identified for the plutonium and uranium isotopes. The results are summarized in Figure D-1

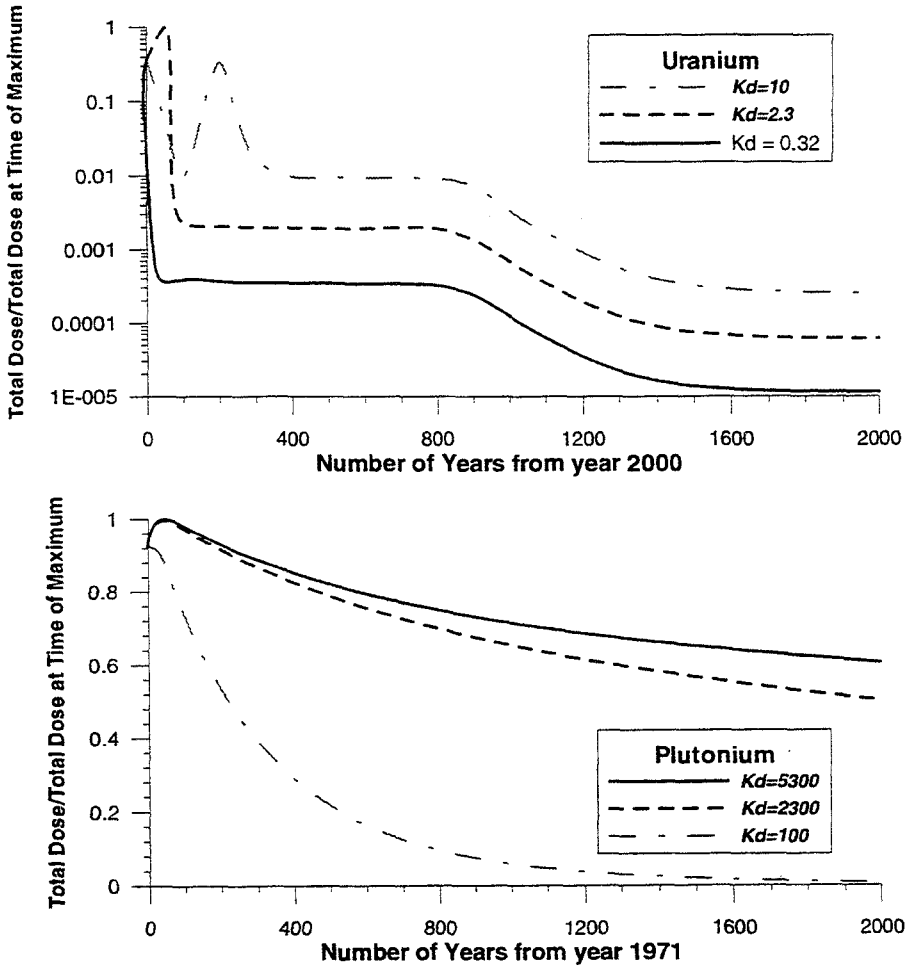

Figure D-1. The ratio of the total dose (all isotopes) for a given year to the maximum total dose for the simulation, as a function of time from the start of the simulation and for several different K_d values. Uranium isotopes include ^{234}U, ^{235}U, and ^{238}U. Plutonium isotopes include ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{241}Am, and ^{237}Np. Doses also include all associated progeny.

Doses for plutonium isotopes (lower graph) are driven by surface exposure pathways (soil, plant ingestion, ground exposure, and inhalation). Therefore, doses are proportional to the amount of activity in the surface soil. The slight increase in dose from year 0 to year 30 is from ingrowth of ^{241}Am in the surface soil. After that, doses drops off exponentially as plutonium activity is leached from the surface soil. Depletion of activity from the surface soil is a function of the water infiltration rate and the distribution coefficient (K_d). Low K_d values result in higher leach rates

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and, therefore, more rapid depletion of the surface soil. Doses from groundwater-dependent pathways were all zero for the 2000-year timeframe considered.

Uranium doses (upper graph) are somewhat more complicated because groundwater-dependent doses were appreciable during the 2000-year timeframe considered. For low K_d values (0.32 mL g^{-1}), activity is rapidly leached from the surface soil and moves quickly through the groundwater. For higher K_d values (2.3 and 10 mL g^{-1}), there is a delay in the arrival of the uranium isotopes in the groundwater. Maximum doses are achieved when uranium contamination is at the receptor well, then fall off quickly as the plume moves downgradient. Doses remain at a more-or-less constant level after passage of the uranium plume and then fall off after 800 years. The flat portion of the curve is caused by doses from radioactive progeny.

APPENDIX E

RISKS ASSOCIATED WITH A 15 MREM ANNUAL DOSE LIMIT

APPENDIX E

RISKS ASSOCIATED WITH A 15 MREM ANNUAL DOSE LIMIT[a]

The limit adopted by DOE/EPA/CDPHE (1996) for radionuclide soil action levels (RSALs) is an individual effective dose of 15 mrem y^{-1}. The risk associated with a uniform whole body exposure from penetrating low-LET external radiation, such as gamma radiation or x-rays, is now recognized to be about 5×10^{-2} Sv^{-1} (or 5×10^{-4} rem^{-1}) (ICRP 1991; EPA 1994; Puskin and Nelson 1995) for a population of all ages. This risk estimate is derived from the epidemiological studies of the atomic bomb survivors and is supported by many other epidemiological studies of different populations in different exposure circumstances. The uncertainty in the risk coefficient has been estimated to range from a factor of 4 (5th percentile) below the nominal value of 5×10^{-2} Sv^{-1} to about twice the nominal value (95th percentile) (i.e., from 1.2×10^{-2} Sv^{-1} to 8.8×10^{-2} Sv^{-1}) (NCRP 1997). In this case, the unit Sv[b] refers to the effective dose as defined in ICRP Publication 60 (ICRP 1991). Using this nominal risk coefficient (5×10^{-2} Sv^{-1}), the lifetime risk associated with the dose limit of 15 mrem y^{-1} is:

$$15 \text{ mrem y}^{-1} \cdot 5 \times 10^{-2} \text{ Sv}^{-1} \cdot 10^{-2} \text{ Sv rem}^{-1} \cdot 10^{-3} \text{ mrem rem}^{-1} = 7.5 \times 10^{-6} \text{ for each year}^{c}.$$

For a 70-year lifetime of exposure at the limit, the lifetime dose would be 1050 mrem (1.05 rem) and the lifetime risk would be 5.25×10^{-4}. This compares with a lifetime risk of 3×10^{-4} cited in the CERCLA requirements (EPA 1997) as corresponding to the annual dose limit of 15 mrem y^{-1}. This difference is because the period chosen by EPA (EPA 1996, p45) for exposure in a lifetime is 30 years, as for all CERCLA exposures, and not 70 years. EPA determined that the 30 year time period is the most appropriate for application to site clean-up efforts. This is based on the fact that 30 years represents the national upper bound (90th percentile) time at one residence as determined from the 1983 survey by the Bureau of the Census.

The range of uncertainty on our estimate of a lifetime risk (70y) of 5.25×10^{-4}, resulting from uniform whole body exposure of 15 mrem y^{-1}, is about 1.3×10^{-4} to 9×10^{-4} (5th to 95th percentile). This range includes both the EPA estimate of lifetime risk for 30 years at 15 mrem y^{-1} (i.e., 3×10^{-4}) and our estimate of lifetime risk for 30 years (i.e., 2.25×10^{-4}).

The same nominal risks would be expected to result from exposure to a radionuclide uniformly distributed in the body, such as tritium, although the uncertainties may be different depending on the exposure circumstances.

Plutonium does not distribute uniformly among the organs and tissues of the body after inhalation, and, consequently, the risk from a given intake is not distributed uniformly in the body either. However, plutonium's distribution after inhalation is comparatively well known. In fact, 97% of the risk arises in only four organs or tissues – lung, liver, bone (i.e. bone surface) and bone marrow, as can be seen from the data on dose per unit intake given in ICRP Publication 71

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[a] Adapted from material provided by W.K. Sinclair, Ph.D.

[b] 1 sievert (Sv) = 100 rem

[c] The actual risk at this dose limit will ostensibly vary with the age of the individual exposed, but over a lifetime the risk will average out at the nominal value of 5×10^{-2} Sv^{-1} because the nominal risk is for a population of all ages.

(ICRP 1995), which is discussed in Grogan et al. (2000). After inhalation of 1-μm AMAD (activity median aerodynamic diameter) particles of ^{239}Pu, the total effective dose is given as 1.6 $\times 10^{-5}$ Sv Bq^{-1} [see ICRP 1995, Table 5.29.3(c) absorption type S, (i.e., slowly absorbed), for adults]. This effective dose is essentially the same as that obtained by adding up the contribution of each of the organs to the effective dose using weighting factors (1.49×10^{-5} Sv Bq^{-1}) (see Table E-1, first 3 columns). Table E-1, column 4, gives the mortality risk per unit dose for each organ (Grogan et al. 2000). Column 5 shows the total risk for each organ for a given ^{239}Pu intake, calculated by multiplying column 1 and column 4 values.

The lifetime risk associated with the effective dose for plutonium is obtained by multiplying it (1.6×10^{-5} Sv Bq^{-1}) by the nominal risk (5×10^{-2} Sv^{-1}) to yield 8.0×10^{-7} Bq^{-1}. Alternatively, and useful as a check, the risk from this effective dose can be obtained by adding up the individual components of the risk in each of the organs or tissues (Table E-1, column 5) calculated using mortality risks per unit dose for these organs and tissues, shown in column 4 (Grogan et al. 2000, Table 9-4). This sums the actual risks for organs and tissues, not using weighting factors, as shown in column 5 of Table E-1. These components of the risk sum to 7.14 $\times 10^{-7}$ Bq^{-1} (Column 5, Table E-1), which is in reasonable agreement with 8.0×10^{-7} Bq^{-1}, especially considering that the risks estimated in Grogan et al. (2000) were newly developed and proposed.

Thus, the total risk of any given intake by inhalation, although distributed very differently, is essentially the same in this case for a given effective dose as for a uniform exposure[d]. The uncertainty, however, is much greater, about a factor of 30 in either direction or a range of about 10^3 for the risk from plutonium (Grogan et al. 2000, Section 9.6) compared with a range of only 8 for the nominal value of risk from external radiation.

E.1 SUMMARY OF RISK ASSOCIATED WITH DOSE LIMIT

In summary, the risk following an effective dose of plutonium of 15 mrem/y for a lifetime of 70 years is about 5×10^{-4} ranging from about 2×10^{-5} to about 15×10^{-3} for a population of all ages (and 4×10^{-4} ranging from 2×10^{-5} to 12×10^{-3} for adults only). For an exposure lifetime of 30 years, as used by the CERCLA program, it is about 2.3×10^{-4} probably ranging from about 1 $\times 10^{-5}$ to about 7×10^{-3}.

[d] The effective dose is often a relatively crude indication of fatal cancer risk since, except in the case of uniform distribution, it depends on the ICRP assigned weighting factors. These are based on fractions of the total health detriment and the detriment includes factors other than the fatal cancer risk. Furthermore the fractions are rounded at that and are sometimes very approximate. This can sometimes lead to discrepancies between the risks obtained by summing the contributions of the risk from the individual organs and the risk apparently associated with the effective dose (see, for example, EPA 1999). As we have seen in the case of plutonium by inhalation the discrepancies are not large especially compared with the uncertainties.

E.3

Table E-1. Contribution of organs to the effective dose and risk per unit intake for ^{239}Pu

	(1) Dose ($Sv\ Bq^{-1}$)	(2) W_T[a]	(3) Contribution to effective dose ($Sv\ Bq^{-1}$) {(1) × (2)}	(4) Mortality Risk[b] (Sv^{-1})	(5) Total mortality risk (Bq^{-1}) {(1) × (4)}
Lung	8.7×10^{-5}	0.12	1.044×10^{-5}	6.55×10^{-3}	56.99×10^{-8}
Liver	3.9×10^{-5}	0.05	0.195×10^{-5}	2.85×10^{-3}	11.12×10^{-8}
Bone	1.8×10^{-4}	0.01	0.18×10^{-5}	0.065×10^{-3}	1.17×10^{-8}
Bone marrow	3.2×10^{-7}	0.12	0.004×10^{-5}	0.65×10^{-3}	0.02×10^{-8}
Skin	3.2×10^{-7}	0.01	0.0003×10^{-5}		69.3×10^{-8}
Ovaries Testes	2.5×10^{-6}	0.2	0.05×10^{-5}		$\times \underline{1.03}$ $= 71.4 \times 10^{-8}\ Bq^{-1}$
Remainder	3.4×10^{-7}	0.05	0.0017×10^{-5}		or
Thyroid	3.2×10^{-7}	0.05	0.0016×10^{-5}		$= 7.11 \times 10^{-6}\ Sv^{-1}$
Colon	3.3×10^{-7}	0.12	0.0040×10^{-5}		
Oesophagus	3.5×10^{-7}	0.05	0.0017×10^{-5}		
Stomach	3.5×10^{-7}	0.12	0.0042×10^{-5}		
Breast	3.2×10^{-7}	0.05	0.0016×10^{-5}		
Sum of organs and tissues[c]			1.49×10^{-5}		

[a] From ICRP(1991)
[b] From Table 9-4, Grogan et al. (2000), but divided by 20 for Sv^{-1}
[c] Whole body = 1.6×10^{-5} Sv Bq^{-1} (according to ICRP 1995, Table 5.29.3(c))

E.2 RISK LIMITS VERSUS DOSE LIMITS

The joint Task Force of the DOE/EPA/CDPHE, after much detailed consideration (DOE/EPA/CDPHE 1996), decided to specify a dose limit, 15 mrem y^{-1}, to which the RSALs must conform rather than a risk limit. This is an important matter for anyone or any organization attempting to define RSALs and/or other specified limiting quantities because of the various, indeed manifold, past discussions and writings of the EPA and other responsible government agencies on the question of specifying risks that the public might reasonably be exposed to. This, of course, assumes that zero risk is not only impractical and impossibly costly in the case of contaminated soils but also counter-productive in that the effort to achieve it can give rise to more serious risks that outweigh those for which remediation is sought (EPA 1996). It is worth considering the background of risk limits versus dose limits and the status of the problem at the present time.

It is completely understandable that an agency such as EPA, given the responsibility for protecting the public against contamination from all kinds of noxious agents, would want to use a risk basis for their limits, especially if all of these agents induce cancer as the risk of primary concern. Risk is then the only common currency as a measure of the effects of physical agents such as ionizing radiation, ultraviolet, ultrasound, etc.; chemical agents such as arsenic, benzene, chloroform, etc.; biological agents such as aflatoxin, etc.; and presumably any other cancer inducing agent, whatever its nature. It also enables the effects of modifiers of these risks such as

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APPENDIX C

SOIL SAMPLING PROTOCOL

FINAL REPORT

Task 6: Sampling Protocols

Radionuclide Soil Action Level Oversight Panel

December 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

"Setting the standard in environmental health"

Risk Assessment Corporation
417 Till Road, Neeses, SC 29107
Phone 803.536.4883 Fax 803.534.1995

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FINAL REPORT

Task 6: Sampling Protocols

Radionuclide Soil Action Level Oversight Panel

December 1999

Contributing Authors

David J. Thorne, Independent Consultant
Arthur S. Rood, K-Spar, Inc.

Principal Investigator

John E. Till, Ph.D., *Risk Assessment Corporation*

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

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EXECUTIVE SUMMARY

The primary goal of Task 6 is to develop recommendations for a soil sampling protocol for use at the Rocky Flats Environmental Technology Site (RFETS) to obtain soil concentration data for comparison to the soil action levels. Sampling protocols are written descriptions of the detailed procedures to be followed in collecting, packaging, labeling, preserving, transporting, and documenting the samples.

Sampling protocols are developed using the iterative data quality objective (DQO) process and require the U.S. Department of Energy (DOE) and its contractor to evaluate several important considerations. These considerations include evaluating sampling and analytical costs in relation to available resources and accepting potential decision errors that may result in remediating sites that are judged contaminated when they are actually below the soil action levels. Conversely, developing a sampling protocol must also incorporate the concerns of the general public and other stakeholders, which are represented by the Radionuclide Soil Action Level Oversight Panel (RSALOP) and the soil action level study. Because of the complexity of developing sampling protocols, with the inherent need to balance the concerns of DOE and the RSALOP, developing a comprehensive sampling protocol was not considered possible. In this report, *Risk Assessment Corporation* (*RAC*) presents recommendations for those elements of a soil sampling protocol considered essential to ensure representative soil samples are collected for comparison to the soil action levels. These recommendations are provided to the RSALOP for presentation to DOE and its contractor, Kaiser-Hill Company, for incorporation into the soil sampling protocol and procedures to be used for the soil action level study.

RAC conducted a review of the current sampling program used at the RFETS and found that a specific sampling protocol for the soil action levels study had not been developed. However, during this review, several procedures were identified that are available in the Rocky Flats program for incorporation into a sampling protocol. Current procedures for packaging, labeling, preserving, transporting, and documenting the samples were considered appropriate for use in a soil sampling protocol for the soil action levels study. Therefore, the main emphasis of the sampling protocol recommendations is directed toward sample collection and sampling designs. This report reviews several methods currently in use at the RFETS for collecting soil samples. The main concern with sample collection is to ensure that representative samples of the surface soil (i.e., 0 to 20-cm depth based on the conceptual model for the soil action levels) are collected. Soil sample collection protocols must be based upon the conceptual model used to derive the soil action levels to ensure that representative soil samples are collected.

This report presents recommendations for a soil sampling protocol to support the final status survey. The final status survey determines the final condition of the site and is performed after decontamination activities are complete. This survey provides the data to demonstrate that radionuclide concentrations in soil satisfy the established soil action levels.

Recommendations for a sampling protocol in support of remedial action were not developed for the Task 6 report. Soil sampling in support of remedial action is an important concept, however, a large number of soil samples have already been collected for use in evaluating the nature and extent of contamination in the surface soil at the RFETS (see the Task 3 report). Several studies detailing the nature and extent of contamination in the surface soil at the RFETS have also been conducted and are available for use in evaluating remedial requirements.

The U.S. Environmental Protection Agency, U.S. Nuclear Regulatory Commission, DOE, and U.S. Department of Defense developed *the Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM)* (NRC 1997), which provides detailed guidance for planning, implementing, and evaluating environmental and facility radiological surveys conducted to demonstrate compliance with dose-based soil action levels. The NRC (1997) guidance focuses on demonstrating compliance based on the final status survey results. NRC (1997) is the most comprehensive guidance document currently available for developing radiological surveys. In this report, *RAC* uses the general principles of the MARSSIM guidance for developing recommendations for a sampling protocol and emphasizes problems with applying the MARSSIM guidance to the soil action levels at Rocky Flats and potential solutions to these problems.

RAC provides several recommendations in this report in support of developing a surface-soil sampling protocol for the final status survey. The following is a general summary of the recommendations with references to the report sections that provide additional detail. *RAC* recommends that

1. The DQO process be used to develop the soil sampling protocol for the final status survey (Section 3.1).

2. DOE appoint representatives from the RSALOP for inclusion on the DQO planning team (Section 3.1).

3. The RSALOP select discrete values from the soil action level distributions, for each radionuclide, and use these discrete values for comparison to the soil concentration data (Section 3.2).

4. Soil samples be collected using profile sampling (Section 4.1).

5. Profile sampling be conducted in soil depth increments of 0–3 cm to be consistent with the resuspension model parameters used to develop the soil action levels (Section 4.1.4).

6. Soil samples should not be composited, rather, individual soil samples should be analyzed for radionuclide contaminants (Section 4.1.6).

7. Soil samples be collected using a systematic grid sampling design, with a random starting point (Section 4.2).

8. A statistician familiar with the RFETS and environmental statistical designs be included on the DQO planning team (Section 4.3).

9. The arithmetic mean of the soil concentration data and its associated uncertainty at the upper 95% confidence interval be used for comparison to the soil action levels (Section 4.3).

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10. The MARSSIM (NRC 1997) non-parametric statistical tests not be used for the soil action level study because these tests compare the median value of the sample distribution to the soil action levels. When the distribution is not symmetrical (i.e., skewed), the median is not equal to the mean (Section 4.3.3).

11. Parametric statistical tests, bootstrapping, or geostatistical techniques be investigated for use in comparing the 95% upper confidence limit of the mean against the soil action levels. The statistical tests should be investigated during the DQO process and chosen according to knowledge of the areas to be sampled (Section 4.3.3).

12. Spatial correlations be investigated to determine their presence in the survey unit of interest and to determine if methods are required to improve the estimate of the mean based on the systematic grid sampling method (Section 4.3.4).

13. The null (Ho) and alternative (Ha) hypothesis are stated as Ho: $\mu \geq$ SAL and Ha: $\mu \leq$ SAL (Section 4.3.5) where μ is the mean soil concentration and SAL is the soil action level.

14. The survey units be classified according to the NRC (1997) scheme and that the size of the survey units be limited accordingly (Section 4.6).

15. In situ gamma spectroscopy measurement be performed to identify potential hot spot locations (Section 4.7).

16. Hot spots identified by soil samples or in situ gamma spectroscopy measurements be investigated further to delineate the size of the hot spot and to determine the upper 95% confidence interval of the mean radionuclide concentrations contained in the hot spot (Section 4.7).

17. That hot spots greater than 100 m^2 with arithmetic mean soil concentrations at the upper 95% confidence interval that exceed the soil action levels be remediated (Section 4.7).

18. That hot spots less than 100 m^2 be area averaged with soil concentrations in a 100 m^2 area and area weighted to determine if the upper 95% confidence interval of the mean soil concentration exceeds the soil action levels and, thus, requires remediation (Section 4.7).

19. The use of surrogate measurements in the final status survey. The ratio of ^{241}Am to ^{239,240}Pu provides a mechanism for the in situ gamma spectroscopy measurement of ^{241}Am to be used to predict the soil concentrations of ^{239,240}Pu (Section 4.8).

20. DOE implement an independent verification survey for the radionuclide soil action level project. This survey should be performed by an independent third party (Section 4.9).

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ACRONYMS

ASME	American Society of Mechanical Engineers
CDPHE	Colorado Department of Public Health and Environment
DOD	U.S. Department of Defense
DOE	U.S. Department of Energy
DQI	data quality indicator
DQO	data quality objective
EPA	U.S. Environmental Protection Agency
GRRASP	(EG&G Rocky Flats) General Radiochemistry and Routine Analytical Services Protocol
HPGe	hyper-pure germanium
MARSSIM	Multi-Agency Radiation Survey and Site Investigation Manual
NRC	U.S. Nuclear Regulatory Commission
QA	quality assurance
QAPjP	quality assurance project plan
QC	quality control
RAC	*Risk Assessment Corporation*
RFETS	Rocky Flats Environmental Technology Site
RSALOP	Radionuclide Soil Action Level Oversight Panel
SOP	standard operating procedure

TASK 6: SAMPLING PROTOCOLS

1. 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 unacceptable radiation doses. 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 recommended soil action levels proposed. A Radionuclide Soil Action Level Oversight Panel (RSALOP) was established and a contractor 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 6 is to develop recommendations for a soil sampling protocol for use at RFETS in support of the soil action level study. Sampling protocols are written descriptions of the detailed procedures to be followed in collecting, packaging, labeling, preserving, transporting, storing, and documenting the samples (Keith 1991). The reader is referred to the EPA's guidance document on "Preparation of Soil Sampling Protocol: Techniques and Strategies" for an elementary discussion on sampling protocols (EPA 1983).

Sampling protocols are developed using the iterative data quality objective (DQO) process and require the U.S. Department of Energy (DOE) and its contractor to evaluate several important considerations. These considerations include evaluating sampling and analytical costs in relation to available resources and accepting potential decision errors that may result in remediating sites that are judged contaminated when they are actually below the soil action levels. Conversely, developing a sampling protocol must also incorporate the concerns of the general public and other stakeholders, which are represented by the RSALOP and the soil action level study. Because of the complexity of developing sampling protocols, with the inherent need to balance the concerns of DOE and the RSALOP, developing a comprehensive sampling protocol was not considered possible. In this report, *RAC* presents recommendations for those elements of a soil sampling protocol considered essential to ensure representative soil samples are collected for comparison to the soil action levels. These recommendations are provided to the RSALOP for presentation to DOE and its contractor, Kaiser-Hill Company, for incorporation into the soil sampling protocol and procedures to be used for the soil action level study.

The primary concern of this report is to develop recommendations for the design of site-specific surface soil sampling procedures that ensure representative samples are collected to determine soil action levels. Soil sampling protocols must be based upon the conceptual model used to derive the soil action levels to ensure that representative soil samples are collected. For example, the depth to which a sample is taken may affect the measured concentration if the radionuclide is deposited in the top few centimeters. Under some circumstances, averaging over the top 15 cm (approximately 6 in.) is appropriate if the exposure pathway of concern is ingestion of food raised in the area. However, averaging may underestimate the potential dose if the exposure pathway of concern is soil ingestion or inhalation of resuspended dust (SAB 1997).

This report presents recommendations for a soil sampling protocol to support the final status survey. The final status survey determines the final condition of the site and is performed after decontamination activities (if required) are complete. This survey provides the data to demonstrate that radionuclide concentrations in soil satisfy the established soil action levels. This survey is also referred to as a termination survey, post remedial-action survey, and final survey.

Recommendations for a sampling protocol in support of remedial action were not developed for the Task 6 report. Soil sampling in support of remedial action is an important concept; however, a large number of soil samples have already been collected for use in evaluating the nature and extent of contamination in the surface soil at the RFETS (see the Task 3 report). Several studies detailing the nature and extent of contamination in the surface soil at the RFETS have also been conducted and are available for use in evaluating remedial requirements. This is not to imply that no further surface soil studies should be conducted at RFETS in support of remedial design. In fact, as noted in this report, additional studies should be conducted for uranium contamination in the surface soil at RFETS. However, *RAC* and RSALOP concurred that the emphasis of this report should be placed on the final status survey.

The soil sampling protocol recommendations presented in this report are based on the conceptual model, parameterization, and assumptions used to develop the soil action levels presented in the Task 5 report. The conceptual model for the soil action levels (Task 5) report is based on the surface soil (0 to 20 cm) layer. Therefore, the recommendations presented in this report are not applicable to subsurface soil layers that may be of concern for evaluating source inventories for use in groundwater transport models. Future work by the Actinide Migration Panel may indicate that groundwater transport and seeps to surface water are important processes at the RFETS. This finding would require that a sampling protocol be developed for application to subsurface soil layers (i.e., > 20 cm).

The EPA, NRC, DOE, and DOD have developed *the Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM)* (NRC 1997), which provides detailed guidance for planning, implementing, and evaluating environmental and facility radiological surveys conducted to demonstrate compliance with dose-based soil action levels. The MARSSIM guidance focuses on demonstrating compliance during the final status survey following scoping, characterization, and any necessary remedial actions. The MARSSIM (NRC 1997) is the most comprehensive guidance document currently available for the development of radiological surveys. DOE and their site contractor, Kaiser-Hill Company, have used the MARSSIM guidance for use in final status surveys of building contamination. Therefore, *RAC* recommends that the final status surveys conducted at RFETS follow the general principles provided in the MARSSIM guidance for the soil action level project. In this report, *RAC* has used the general principles of the MARSSIM (NRC 1997) guidance to develop recommendations for a sampling protocol. The MARSSIM guidance and methods should not be used blindly as a recipe for final status surveys. *RAC* provides an emphasis in this report on problems identified with the MARSSIM guidance in terms of application to the soil action levels at Rocky Flats and potential solutions.

2. REVIEW OF THE ROCKY FLATS SOIL SAMPLING PROGRAM

RAC conducted a review of the current sampling program and procedures used at the RFETS to determine elements of the current program available for use and incorporation into the final status sampling protocol. We reviewed quality assurance (QA) requirements, standard operating procedures (SOPs), and individual site sampling and analysis plans. This section summarizes the sampling program and procedures currently used at Rocky Flats.

A soil sampling protocol for specific application to the soil action levels was not identified during this review. Because a soil sampling protocol directly addressing the sampling requirements and associated data needs for comparison to the soil action levels has not been developed for use at Rocky Flats, this review makes no attempt to evaluate the quality of such a sampling program. The review was conducted to identify the program elements and procedures available for use in developing a soil sampling protocol for the soil action levels.

RAC identified several procedures currently used at the RFETS that can be used to develop a soil sampling protocol. Overall, the administrative and field procedures were considered to be technically sound and based on standard industry guidelines. However, soil sampling methods available at Rocky Flats were not considered adequate for ensuring that representative samples are collected for the soil action level project.

The documented sampling program at the RFETS is based on the present guidelines and requirements of the American Society of Mechanical Engineers (ASME) NQA-1 (ASME 1989); DOE Order 5700.6c (DOE 1991); DOE Order 5400.1 (DOE 1989); EPA guidance for Comprehensive Environmental Response, Compensation, and Liability Act sites (including the DQO process) (EPA 1994a); and EPA, NRC, DOE, and DOD's *Multi-Agency Radiation Survey and Site Investigation Manual (MARSSIM)* methodologies (NRC 1997). Soil sampling programs at the RFETS are largely based upon EPA's Comprehensive Environmental Response, Compensation, and Liability Act guidance, which has been adopted, in part, by NRC (1997).

2.1 Quality Assurance Project Plan

The overall soil sampling QA program document for the RFETS is contained in the quality assurance project plan (QAPjP), which is titled *Rocky Flats Plant Environmental Management Site-Wide QA Project Plan* (EG&G Rocky Flats 1994a). This QAPjP meets the requirements set forth in the following guidance and regulatory documents:

- EPA QAMS/005/80, *Interim Guidelines and Specifications for Preparing Quality Assurance Project Plans* (EPA 1980a)
- DOE Order 5700.6c, "Quality Assurance" (DOE 1991)
- DOE Order 5400.1, "General Environmental Protection Program" (DOE 1989)
- ASME NQA-1, *Quality Assurance Program Requirements for Nuclear Facilities* (ASME 1989).

The QAPjP describes the policy, organization, functional responsibilities, and QA requirements for programs at RFETS. In addition to the QAPjP, SOPs at Rocky Flats describe the field techniques to be used during soil sampling field investigations at the RFETS. The SOPs, together with the QAPjP, form the RFETS sampling and analysis plan.

In addition to the QAPjP and SOPs, the QAPjP requires developing operable unit-specific work plans or field sampling plans. Operable unit is a term used by the EPA to identify specific contamination areas to be considered as one unit for an assessment. Work plans describe how each operable unit will be characterized and include specific operable unit background information, sampling objectives, sample locations, and minimum frequency for each task or operation.

The QAPjP also requires that each work plan be accompanied by a QA addendum. The QA addendum outlines the site- or operable unit-specific measures to be taken to meet the QA requirements in the QAPjP and references the SOPs to be followed during the investigation of a specific operable unit. In addition, specific SOPs may be developed for desired variations in the standard SOPs that are necessary for specific tasks in a particular operable unit.

The QAPjP also provides guidance on QA for soil sample data quality indicators (DQIs). Table 2-1 gives the minimum frequency requirements set forth in the QAPjP.

Table 2-1. Quality Control Sample Minimum Collection Frequency at the RFETS

Activity	Frequency
Field duplicate	1 in 20
Field blanks[a]	As specified in work plan/QA addendum
Trip blanks	As specified in work plan/QA addendum
Equipment rinse blanks	1 in 20 or once per day, whichever is more frequent
Other quality control activities	As specified in work plan/QA addendum

[a] According to the QAPjP, the use of field blanks for soil and sediment sampling at the RFETS is not appropriate because of the lack of commercially available blank soil and solid materials that adequately reflect the various soil types encountered. Developing blank soil types within the Rocky Flats Plant region is not practical because of the subjectivity of characterizing background soil conditions and variability of soil types.

2.2 RFETS Standard Operating Procedures

The SOPs at the RFETS for soil sampling are contained in two main procedural documents: (1) *Environmental Management Administrative Procedures*, Manual No. 3-21000-ADM and (2) *Environmental Management Division (EMD) Operating Procedures*, Manual No. 5-21000-OPS.

2.2.1 Environmental Management Administrative Procedures

The EG&G Rocky Flats *Environmental Management Administrative Procedures,* Manual No. 3-21000-ADM, contains administrative-level procedures. These procedures provide the requirements for developing QA addenda, procedures, forms, and records management. Table 2-2 identifies procedures that are directly related to the topic of soil sampling at the RFETS.

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Table 2-2. RFETS Administrative Standard Operating Procedures Related to Soil Sampling

Procedure number	Procedure title
3.04	Control of Quality Assurance Addenda (QAA) Development
5.01	Procedure Development
5.03	RCRA Facility Investigation (RFI)/Remedial Investigation (RI) Work Plan Development
5.08	Forms Control
6.01	Document Control
8.01	Control and Identification of Samples and Data
17.01	Quality Assurance Records Management

2.2.2 Environmental Management Operating Procedures

Environmental Management Division (EMD) Operating Procedures, Manual No. 5-21000-OPS, consists of several volumes of SOPs. Volumes that are pertinent to soil sampling include *Volume I: Field Operations*, Manual No. 5-21000-OPS-FO, and *Volume III: Geotechnical*, Manual No. 5-21000-OPS-GT. These SOPs are provided to the field personnel and describe the procedural steps required to complete a specific task. Table 2-3 lists the procedures in these volumes that are pertinent to soil sampling at the RFETS.

Table 2-3. RFETS Standard Operating Procedures Related to Soil Sampling

Procedure number	Procedure title
FO.03	General Equipment Decontamination
FO.10	Receiving, Labeling, and Handling Environmental Materials Containers
FO.13	Containerization, Preserving, Handling and Shipping of Soil and Water Samples
FO.18	Environmental Sample Radioactivity Content Screening
GT.08	Surface Soil Sampling
GT.17	Land Surveying

2.3 RFETS Analytical Laboratory Requirements

The *EG&G Rocky Flats General Radiochemistry and Routine Analytical Services Protocol* (GRRASP) provides the procedures for analytical laboratory work at Rocky Flats (EG&G Rocky Flats 1994b). Technical requirements in the GRRASP specify the methods to be used, required detection limits, and deliverables necessary.

The analyses of radionuclides at the RFETS are conducted in accordance with the standards and guidance set forth in the following documents:

- *Test Methods for Evaluating Solid Waste* (EPA 1986)

- *Radiochemical Analytical Procedures for Analysis of Environmental Samples* (EPA 1979)
- *Interim Radiochemical Methodology for Drinking Water* (EPA 1976)
- *Prescribed Procedures for Measurement of Radioactivity in Drinking Water* (EPA 1980b)
- *Standard Methods for the Examination of Water and Wastewater* (APHA 1989)
- *Eastern Environmental Radiation Facility Radiochemistry Procedures Manual* (EPA 1984)
- *Methods for Chemical Analysis of Water and Wastes* (EPA 1974)
- *Procedures for Radiochemical Analysis of Nuclear Reactor Aqueous Solutions* (EPA 1973)
- *The Procedures Manual of the Environmental Measurements Laboratory* (DOE 1997)
- U.S. Nuclear Regulatory Commission Regulatory Guides.

Laboratory method detection limits and DQOs are provided in Appendix B of the QAPjP.

2.4 RFETS Data Validation Requirements

Functional guidelines for validating most radiochemistry data have not been published by EPA; however, data validation functional guidelines, applied directly from EPA Contract Laboratory Program, have been established for the RFETS. The functional guidelines used to validate analytical data at the RFETS include

- *Laboratory Data Validation Functional Guidelines for Evaluating Organic Analyses* (EPA 1988a)
- *Laboratory Data Validation Functional Guidelines for Evaluating Inorganic Analyses* (EPA 1988b)
- *Water Quality Parametric Data Validation Guidelines* (EG&G Rocky Flats 1990a)
- *Radiochemical Data Validation Guidelines—Tritium Analysis by Liquid Scintillation* (EG&G Rocky Flats 1990b)
- *Radiochemical Data Validation Guidelines—Isotopic Analyses by Gamma Spectroscopy* (EG&G Rocky Flats 1991)
- *Radiochemical Data Validation Guidelines—Gross Alpha/Beta by Gas Proportional Counters* (EG&G Rocky Flats 1990c)
- *Radiochemical Data Validation Guidelines—Isotopic Analyses by Alpha Spectroscopy* (EG&G Rocky Flats 1990d).

Laboratory quality control (QC) procedures are in place for radiochemistry. The laboratory QC procedures and samples used are described in detail in the analytical methods cited in the GRRASP.

2.5 RFETS Soil Sample Collection Protocols

Soil sample collection protocols used at the RFETS are described in EMD Operating Procedure GT.08, Manual No. 5-21000-OPS-GT, Volume III: Geotechnical. This operating procedure describes the surface (near-surface) soil sampling procedures in use at the RFETS.

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Near-surface soil is defined in the SOP as those soils between the ground surface and 1 m (3.3 ft) in depth. The procedure states that

> The purpose of surface soil sampling at the RFETS can be related to one or more specific objectives. These are as follows: 1) resuspension availability, which determines if radionuclides are present in the top-soil that could become resuspended in the air and thus pose a migration pathway by inhalation; 2) deposit inventories, which determine the amount of accumulated radionuclides deposited on the ground; 3) distribution of contaminants, which defines the areal distribution of contaminants; and 4) deposition increment, which defines the areal distribution with depth of radionuclides in the top 15 cm (6 inches) of soil to verify the results of the Hyper-pure germanium (HPGe) surveys.

To meet these objectives, four radionuclide, surface soil sampling methods are used at the RFETS: (1) CDPHE method, designed to sample for resuspension availability, (2) Rocky Flats method, designed to sample for deposited inventories, (3) grab sampling (spade and scoop) method for under asphalt or concrete or where contamination may have occurred from a given point source, designed to sample for contaminant distribution, and (4) vertical soil profile method, designed to sample for deposition increment.

2.5.1 Colorado Department of Public Health and Environment Method

The CDPHE sampler is one method for collecting soil samples for radionuclide analyses at the RFETS. The CDPHE sampler was designed to sample radionuclides in the topsoil that could become resuspended in the air and, thus, pose a migration pathway and exposure via inhalation. The sampler was designed to obtain a sample from the upper surface ¼ in. deep, from an area 2 in. wide and 2-3/8 in. long. Figure 2-1 is a drawing of the CDPHE soil sampling device.

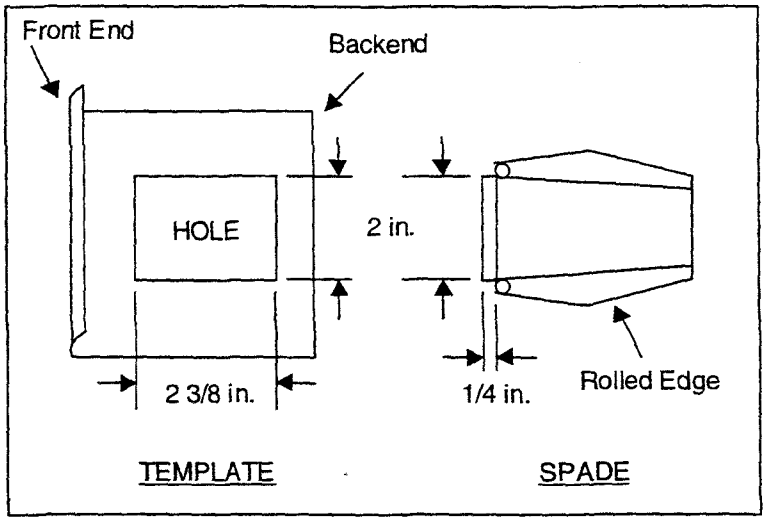

Figure 2-1. CDPHE soil sampling device. The sampler is designed to sample from the upper ¼ in. of soil.

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Vegetation and any undesirable top layer of surficial material are removed. The CDPHE template is placed on the ground so that the soil surface is even with the upper surface of the sampling template. The sampling spade is placed at the backend of the template, and the front edge of the spade is placed into the soil ¼ in. deep. The sampling spade is drawn toward the front of the template to obtain the surface soil sample. The spade is again placed in the sampling hole and drawn forward to ensure that the ¼ in. soil sample has been obtained. The soil sample is then placed into a sample container.

The total number of samples and their locations are specified in the site-specific field sampling plans for each project. A specific number of samples are collected and composited from within each sample plot. The sample locations are described by an evenly spaced grid.

2.5.2 Rocky Flats Method

The Rocky Flats method is used to determine the amount of accumulated plutonium that has been deposited on the ground. This determination is accomplished by collecting a sample volume of 5000 cm^3 of soil in situ. The Rocky Flats jig outlines a square area with 10-cm sides and is driven 5 cm into the soil to cut three sides of the sampler (see Figure 2-2). At the fourth side, soil is removed from outside the jig's perimeter. The scoop is used to finish the cut on the fourth side (open face) of the sample and the bottom surface.

Ten samples are collected at each location and composited. These 10 samples are collected at the center and corners of two 1-m squares that are spaced 1 m apart. Figure 2-3 illustrates this sample collection spacing. The soil samples are passed through a 10-mesh metal sieve to remove large particles (such as cobbles and stone) that do not pass through the sieve. After sieving, the 10 soil samples are composited, mixed, and quartered to obtain a sample for laboratory analysis.

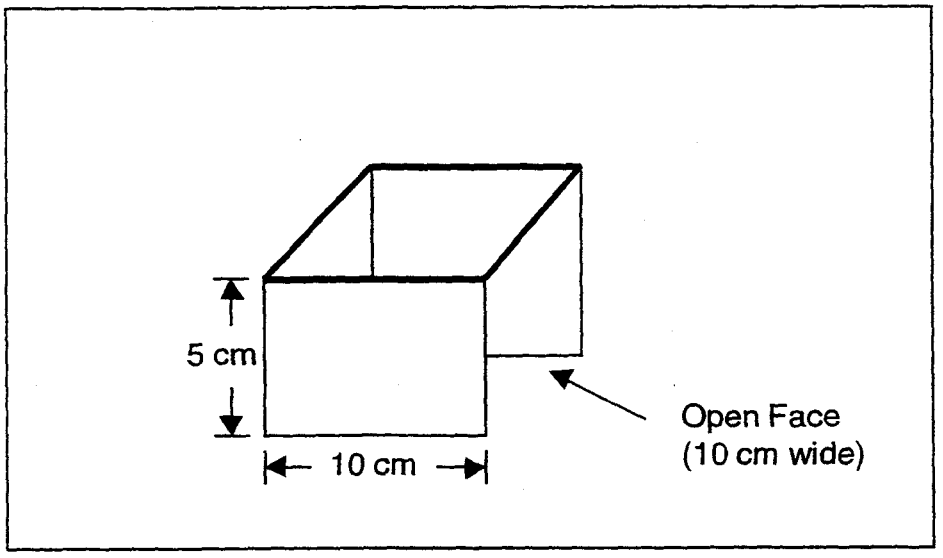

Figure 2-2. Rocky Flats soil sampling device. The device is designed to sample a 5 cm depth.

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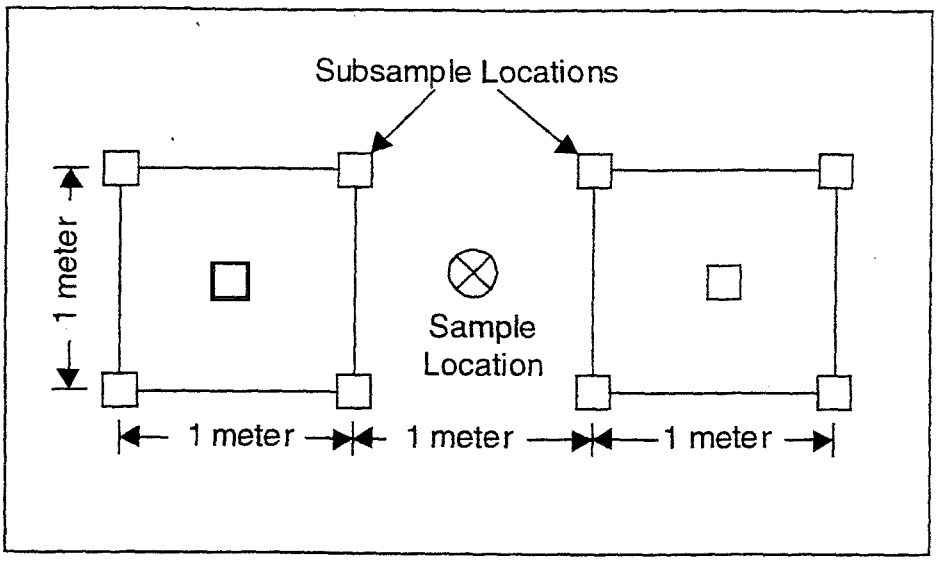

Figure 2-3. Rocky Flats soil sampling location spacing.

2.5.3 Grab Sampling Method (Spade and Scoop)

Surface soil samples can be collected for radionuclide analyses using grab sampling (spade and scoop) methods at RFETS. The vegetation and any undesired surficial material are removed from the area to be sampled. The soil sample is then collected to the desired depth using a stainless steel spoon or scoop.

The total number of samples and sample locations are specified in the site-specific field sampling plans.

2.5.4 Vertical Soil Profile Method

The vertical soil profile sampling at the RFETS defines the distribution of radionuclides in the top 6 in. of soil to verify the results of the HPGe surveys. This sampling is accomplished by collecting discrete soil samples at 2-in. intervals corresponding to depths from 0–2 in., 2–4 in., and 4–6 in. Sampling of 2 in. is required to define the extent of radiological contamination within discrete layers of the surface soil. Four RFETS procedures are used to obtain these samples: (1) collection from the surface downward, (2) collection from the side wall of a small excavation, (3) collection by coring, and (4) collection from beneath concrete and asphalt pavement. The total number of samples to be collected and their locations are specified in site-specific field sampling plans. A sample of approximately 500 g is obtained for each soil profile interval.

Soil sampling is required for in situ gamma spectroscopy surveys using HPGe detectors. In general, the depth distribution needs to be investigated to ensure that the correct parameters are used in the conversion from instrument count rate to soil concentration. As an example (from NUREG-1506, *Measurement Methods for Radiological Surveys in Support of New Decommissioning Criteria, Draft Report for Comment* [NRC 1995a]), for undisturbed soils a negative exponential profile with depth has frequently been found to be an adequate model for deposited radionuclides, that is

$$S = S_0 e^{-\frac{\alpha}{\rho} z\rho} \tag{2-1}$$

where S is the activity per unit volume of soil (pCi cm^{-3}) at depth z (cm), S_o is the activity per unit volume at the soil surface (pCi cm^{-3}), α is the reciprocal of the relaxation length of the exponential distribution (cm^{-1}), and ρ is the bulk density (g cm^{-3}). This expresses the profile in terms of the soil mass per unit area, ρz (g cm^{-2}), with the degree of penetration into the soil represented by the depth parameter α/ρ (cm^2 g^{-1}). This type of profile has the maximum concentration at the soil surface (S_o) and decreases with depth. If the value of α/ρ approaches infinity, the source distribution approaches a plane atop the ground, and if α/ρ equals 0, the source distribution is uniform with depth. As an example, assume a soil density of 1.5 g cm^{-3} and an α/ρ value of 0.2 cm^2 g^{-1} (which is a typical value for an aged fallout deposit). The corresponding relaxation depth for the exponential profile would be 3.33 cm, meaning that the concentration would be reduced to 1/e or 37% of the surface value at this depth. For in situ measurements, the value of α/ρ can be determined from the analysis of soil samples from different depth increments. The fraction of the total activity below a given depth (log value) can be plotted versus the mass depth, ρz. The slope of the line is then the value of α/ρ.

2.6 Summary of RFETS Soil Sampling Program Review

RAC conducted a review of the current sampling program and procedures used at the RFETS to determine elements of the current program available for use and incorporation into the final status soil sampling protocol. *RAC* identified several procedures currently used at the RFETS that can be used to develop a soil sampling protocol. Overall, the administrative and field procedures were considered to be technically sound for their intended purpose and based on standard industry guidelines.

Field procedures have been developed for use at the RFETS that provide field personnel with the required documentation for performing assigned field duties and studies. Standard procedures are in place that document the steps necessary to containerize, label, and ship samples to the laboratory. In addition, the procedures address such issues as equipment decontamination, sample location determination using standard survey and global positioning systems, and documentation of the survey process, including sample chain of custody. These procedures are considered to be technically sound and acceptable for use in a soil sampling protocol for the soil action levels.

Four protocols for surface soil sample collection used at Rocky Flats are discussed in Section 2.5. *RAC* does not consider these collection protocols adequate for ensuring the collection of representative samples for comparison to the soil action levels. Therefore, this report places a major emphasis on recommendations for collecting representative samples for comparison to the soil action levels.

3. SAMPLING DESIGN DEVELOPMENT PROCESS

This section discusses the process to develop soil sampling studies and associated protocols. The MARSSIM recommends the use of the DQO process for the planning, development, and implementation of radiological surveys (NRC 1997).

3.1 Data Quality Objective Process

The process of planning a soil sampling study, implementing the sampling plan, and assessing the sampling results before making a decision is called the data life cycle. Soil sample survey planning uses the DQO process to ensure that the sampling results are of sufficient quality and quantity to support the final decision.

3.1.1 Data Quality Objective Process Description

The DQO process is described in detail in EPA (1994a) and NRC (1997). The DQO process consists of seven steps as shown in Figure 3-1.

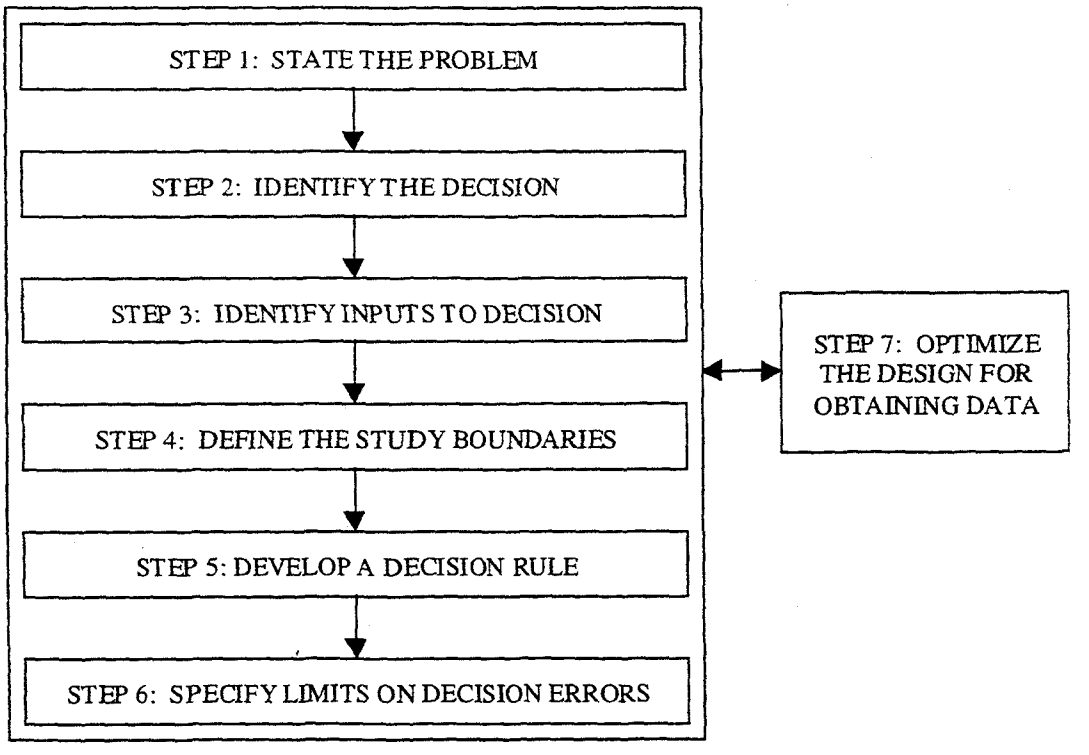

Figure 3-1. The data quality objectives process (from EPA 1994a).

Although the DQO process appears to be linear, the actual process is iterative. During decisions in the process, DQOs in previous steps may need to be reconsidered or redefined. This iteration is encouraged because it ultimately leads to a more efficient survey design (NRC 1997).

Planning radiological surveys using the DQO process can improve the survey effectiveness and efficiency and, thereby, the defensibility of decisions. It also can minimize data collection expenditures by eliminating unnecessary, duplicative, or overly precise data. The use of the DQO process assures that the type, quantity, and quality of environmental data used in decision-making will be appropriate for the intended application. It provides systematic procedures for defining the criteria that the survey design should satisfy, including when and where to perform measurements, the level of decision errors for the survey, and how many measurements to perform (NRC 1997).

The following discussion provides a brief introduction to the DQO process. The reader is referred to the MARSSIM (NRC 1997) for a full discussion on the DQO process and its use in survey planning.

Step 1. State the Problem

The first step in any decision-making process is to define the problem so that the focus of the survey will be unambiguous. Four activities are associated with this step:

- Identifying members of the planning team and stakeholders
- Identifying the primary decision maker or decision-making method
- Developing a concise description of the problem
- Specifying available resources and relevant deadlines for the study.

The expected outputs of this step are

- A list of the planning team members and identification of the decision-maker
- A concise description of the problem
- A summary of available resources and relevant deadlines for the survey.

The planning team clarifies and defines the DQOs for a site-specific survey. This multidisciplinary team of technical experts offers the greatest potential for solving problems when identifying important aspects of a survey. Including representatives from stakeholder groups is an important consideration when assembling this team. Once formed, the team can also consider the role of public participation for this assessment and the possible survey to follow.

A concise description of the problem must be specified during this step. A description of the problem would typically involve the release of all or some portion of the RFETS that demonstrates compliance with the soil action levels. The resources and deadlines for the surveys would need to be addressed by DOE and their site contractor, Kaiser-Hill Company.

Step 2. Identify the Decision

This step defines the question that the survey will attempt to resolve and identify alternative actions that may be taken based on the outcome of the survey. The combination of these two elements is called the decision statement.

Four activities are associated with this step in the DQO process:

- Identifying the principal study question
- Defining the alternative actions that could result from resolution of the principal study question

- Combining the principal study question and the alternative actions into a decision statement
- Organizing multiple decisions.

The expected output from this step is a decision statement that links the principal study question to possible solutions to the problem.

Step 3. Identify the Inputs to the Decision

This step of the DQO process determines and specifies the information needed for the study and data to support a decision concerning the survey unit. Collecting data or information is necessary to resolve the decision statement. For the final status survey, the list of information inputs will involve measurements of the radioactive contaminants of concern in the soil.

The activities included in this step of the DQO process include

- Identifying the information required to resolve the decision statement
- Determining the sources for each item of information
- Confirming that appropriate measurement methods exist to provide the necessary data.

Step 4. Define the Boundaries of the Study

During this step, the DQO planning team develops a conceptual model of the site based on existing information. The conceptual model for the soil action levels has been developed and documented by *RAC* in the Task 2, 3, and 5 reports. The conceptual model is used by the DQO planning team to define the spatial and temporal boundaries of the sampling study. It is very important that the conceptual model and assumptions used to develop the soil action levels are considered during the DQO process.

During this step, the spatial boundaries that will be covered by the decision statement are defined. These considerations include spatial boundaries

- That define the physical area under consideration for release
- That define the physical area to be studied and locations where measurements could be performed
- Developed from modeling used to calculate the soil action levels.

For the final status survey, the smallest, most appropriate subsets of the site for which decisions will be made are defined as survey units (see Section 4.6.4).

Step 5. Develop a Decision Rule

This step defines the statistical parameter of interest, specifies the soil action levels, and integrates previous DQO outputs into a single statement that describes a logical basis for choosing among alternative actions (NRC 1997).

Three activities are associated with this step:

- Specifying the statistical parameter that characterizes the parameter of interest
- Specifying the soil action levels for the study

- Combining the outputs of the previous DQO steps into an "if, then" decision rule that defines the conditions that would cause the decision-maker to choose among alternative actions.

The expected outputs of this step are

- Specification of the parameter of interest that characterizes the level of residual radioactivity
- Specifications of the soil action levels
- An "if, then" statement that defines the conditions that would cause the decision maker to choose among alternative actions.

The parameter of interest is a descriptive measure (such as a mean or median) that specifies the characteristic or attribute of the residual contamination in the survey unit. The specification of the parameter of interest that characterizes the level of residual radioactivity is very important for the soil action level project. The technical basis and recommendation for the statistical parameter of interest is discussed in detail in Section 4.3.

Step 6. Specify Limits on Decision Errors

The following steps are involved in setting acceptable probabilities for decision errors (EPA 1992):

- Defining false positive and false negative errors for the decision and describing the consequences of each type of error
- Evaluating these consequences according to the relative level of concern they would cause, with emphasis on the environment, public health, economics, and social consequences
- Determining if false positive or false negative errors are of greater concern
- Establishing, with the assistance of a statistician, an acceptable probability for the occurrence of each of these errors
- Combining the probability statement into a formal statement of the levels of uncertainty that can be tolerated in the results
- Reviewing the decision rule.

During this step of the DQO process, the null and alternative hypothesis must be chosen along with acceptable probabilities of decision errors. Hypotheses and decision errors are address in Sections 4.3.4 and 4.3.5 of this report.

Step 7. Optimize the Design for Obtaining Data

This step is designed to produce the most resource-effective survey design that is expected to meet the DQOs. It will be necessary for the DQO planning team to work through this step more than once after revisiting previous steps in the DQO process.

Six activities are included in this step:

- Reviewing the DQO outputs and existing data to ensure they are internally consistent
- Developing general data collection design alternatives

- Formulating the mathematical expressions needed to solve the design problem for each data collection design alternative
- Selecting the optimal design that satisfies the DQOs for each data collection design alternative. If the recommended design will not meet the limits on decision errors within the budget or other constraints, then the planning team will need to relax one or more constraints, such as
  - Increasing the budget for sampling and analysis
  - Increasing the decision error rates
  - Changing the boundaries
  - Evaluating alternative measurement techniques with lower detection limits or lower survey costs
  - Considering the use of passive controls when releasing a survey unit rather than unrestricted release
- Selecting the most resource-effective survey design that satisfies all the DQOs
- Documenting the operational details and theoretical assumptions of the selected design in the QAPjP, the field sampling plan, and sampling and analysis plan.

Section 4.2 presents recommendations on the sampling design and mathematical expressions.

3.1.2 Recommendations

- *RAC* recommends that Kaiser-Hill Company, current operating contractor of the RFETS for DOE, use the DQO process as identified in their sampling program and QAPjP to develop the final status soil sampling protocol.

- *RAC* recommends that DOE appoint representatives from the RSALOP for inclusion on the Kaiser-Hill Company DQO planning team. This is an important recommendation to ensure that stakeholder concerns are addressed in developing the sampling protocol and an acceptable sampling protocol for the final status survey is developed. The appointment of RSALOP representatives on the DQO planning team will also ensure that Kaiser-Hill Company has the input of representatives that were directly involved in developing the soil action levels. The inclusion of RSALOP representatives on the DQO planning team is necessary to ensure that decisions that may result in deviations from the recommendations provided in this report include the technical input and acceptance of stakeholders and the general public, represented by the RSALOP.

3.2 Soil Action Levels

3.2.1 Development Method

RAC is conducting the pathway modeling for the radionuclide soil action levels considering parametric uncertainties. In other words, uncertainty in the values of specific model parameters are considered by assigning a probability distribution to each parameter that is treated as uncertain. Parametric uncertainty is concerned with propagating uncertainty in parameter values through the simulations to the resulting estimates of concentrations in exposure media or to dose, typically

using Monte Carlo techniques. The procedure produces an uncertainty distribution for each soil action level.

As discussed in the Task 2 report, the soil action levels must be combined with measured soil concentrations of the respective radionuclides, forming ratios (soil concentration divided by soil action level). If the concentration of radionuclide *i* is denoted by C_i, and its radionuclide soil action level is denoted by SAL_i, then the sum of ratio rule for *n* radionuclides states that

$$\frac{C_1}{SAL_1} + \frac{C_2}{SAL_2} + \frac{C_3}{SAL_3} + \ldots + \frac{C_n}{SAL_n} \leq 1 \quad . \tag{3-1}$$

There are two methods for evaluating the sum of ratios quantity: (1) deterministically and (2) stochastically. In the deterministic case, a discrete soil action level is chosen by the RSALOP for each radionuclide, then the ratios of the soil sample data and the soil action levels are summed in the deterministic case. The sum must be less than or equal to 1 to meet the dose standard (i.e., release standard of 15 or 85 mrem y^{-1}). In the stochastic case, *RAC* has calculated the probability of exceeding the dose limit as a function of the ^{239}Pu soil concentration (Figure 3-2). The doses calculated in the stochastic case include not only the dose from ^{239}Pu, but all other plutonium isotopes and americium. The activity levels of the other plutonium isotopes and americium are estimated using site-specific isotopic ratios described in the Task 5 report (Killough et al. 1999). For example, using Figure 3-2 and a ^{239}Pu soil concentration of 100 pCi g^{-1}, the probability of exceeding the 15 mrem dose limit is about 0.7 or about 70%. We note here that action levels for uranium isotopes have been reported separately because ratios of uranium to plutonium are not consistent across the site. In addition, uranium contamination does not appear to be as widespread as that of plutonium and is mostly restricted to source areas such as old disposal areas and burn pits. (see Section 4.1.3 for further discussion on the distribution of uranium in soil). Soil action levels for ^{238}U are reported in the same manner as those for plutonium isotopes, but they only include doses from the other uranium isotopes and do not include doses from plutonium isotopes. Specifying an action level for a site with both uranium and plutonium contamination will need to be considered case by case.

The *RAC* Task 2 report also provides a discussion about remedial strategies (see Section 3.1.2 and equations in Section 2.1). The Task 2 report notes that programs, such as RESRAD, proceed on the assumption of a uniformly contaminated area (subject to variation within a factor of 3). For some scenarios it could be desirable to subdivide the site area into some number P of plots, each of which can be treated as having a uniform concentration of each radionuclide but with concentrations varying from one plot to another. Such subdivision might be of assistance in the planning for remediation because the effects of reducing the most contaminated plots by various amounts can be studied explicitly. DOE and the Kaiser-Hill Company team are encouraged to consider the use of the subdivision method for evaluating remedial strategies.

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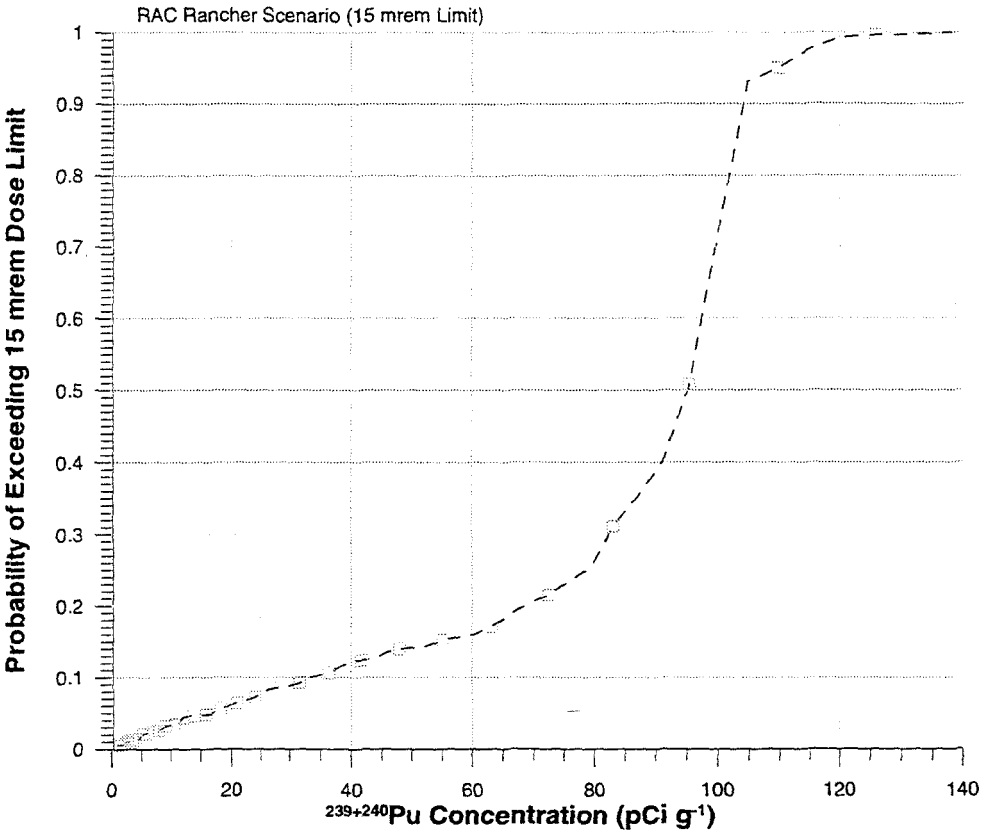

Figure 3-2. Example of a stochastic soil action level that shows the probability of exceeding the 15 mrem dose limit as a function of $^{239+240}Pu$ concentration in soil. Doses include all plutonium isotopes and americium. Activity levels of the other plutonium isotopes and americium are based on site-specific isotopic ratios.

3.2.2 Recommendations

- *RAC* recommends that the RSALOP select discrete values for the soil action levels of each radionuclide for use in the final status survey. Selecting discrete values from the soil action level distributions provides the mechanism for the RSALOP to determine the level of risk considered acceptable for use in the soil action level project.

- *RAC* has provided a potential method for evaluating remedial strategies in the Task 2 report, as described above. The RSALOP, DOE, and the site contractor (Kaiser-Hill Company) are encouraged to consider the use of these methods for evaluating remedial strategies.

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4. SAMPLING PROTOCOL RECOMMENDATIONS

The discussion in Section 3 provides the reader with an understanding of the process required for developing a sampling design and explains the iterative nature of the process. The following discussion provides the recommendations considered by *RAC* to be essential to ensuring that representative surface soil samples are collected for comparison to the soil action levels.

4.1 Soil Sample Collection Protocol

The soil sample collection protocol must ensure that representative samples are collected. A representative sample is a sample collected to appropriately reflect the media and contamination being measured. The samples must also be collected such that the resulting data can be compared to the modeled soil action levels. This requires that the media be sampled in a manner that is consistent with the conceptual model and associated assumptions used in developing the soil action levels. For example, resuspension models are based on contamination contained in the upper soil surface. Sample collection protocols that collect a continuous soil sample from the surface to several centimeters in depth may not be representative of the resuspension model used to derive the soil action levels.

Several soil sample collection protocols (see Section 2.5) are currently used in studies at the RFETS. Several studies on the nature and extent of radionuclide contamination at the RFETS have used either the CDPHE or the Rocky Flats sampling methods, and in some instances, both methods were employed. These two methods, involving different sample collection depths, have created problems for researchers attempting to compare data sets. In addition to these methods, Colorado State University has developed a soil sample collection protocol for use in their radionuclide studies at the RFETS (Webb et al. 1997; Webb 1992), which involves collecting soil samples in 3-cm increments (see Section 4.1.2).

4.1.1 Protocol Comparison Studies

Bernhardt et al. (1983) documents a field soil sampling study conducted around the RFETS in May 1977. The cooperative plutonium soil sampling project was conducted by Rockwell International (contractor at the time for the DOE RFETS); CDPHE; Jefferson County Department of Health; and the EPA Office of Radiation Programs, Las Vegas. Each of the agencies collected five replicate samples from four distinctly different pedological and morphological settings around the RFETS. The following soil sampling methods were used in the study: Rocky Flats method (100 cm^2 area), CDPHE method (750 cm^2 area), Jefferson County technique (sizing of dust swept from a 4 m^2 area of the ground surface), and two EPA methods for samples of 0 to 1 cm (450 cm^2) depth and 0 to 5 cm (500 cm^2) depth.

Bernhardt et al. (1983) found statistically significant differences in average concentration between the sampling techniques. They found that for concentrations expressed on a per gram basis, those sampling methods that sample to greater depths tend to have lower average concentrations. This was attributed to the residence of most of the plutonium in the surface 1 or

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2 cm, and sampling to a greater depth resulted in dilution of the surface soil plutonium with soil from a greater depth that had a lower concentration of plutonium.

Bernhardt et al. (1983) noted that the differences in concentrations observed in the study emphasize the importance of matching sampling objectives with sampling techniques. If the objective is to assess the amount of the total deposited material, samples should be taken to a depth sufficient to collect all the material. If the objective is to assess the health significance of deposited plutonium or to obtain estimates of potential resuspension, samples of only the surface soil should be taken.

Litaor et al. (1995) used both the CDPHE and Rocky Flats methods for sampling of $^{239,240}Pu$ in the Rocky Flats environment. This study noted that because the CDPHE sampler collects only the top 0.64 cm of the soil, the sampler exhibited a serious problem in locating the boundary between the soil surface and the plant litter layer accumulated above. The study concluded that there was no significant difference in $^{239,240}Pu$ activity in soil collected with the CDPHE sampler versus the Rocky Flats sampler for use in developing surface contour maps of estimated plutonium concentrations. Twenty-five samples were composited for the CDPHE method, with 10 samples composited for the Rocky Flats method using survey unit sizes of 1.01 ha near the 903 Pad and 4.05 ha further from the 903 Pad. The Rocky Flats sampling design consisted of taking five subsamples collected from the corners and the center of two 1-m squares, which were spaced 1 m apart in the middle of each survey unit. The CDPHE sampling design required 25 equally spaced subsamples to be composited within each survey unit.

The Litaor et al. (1995) study sampling protocols required composited samples, which introduced an additional source of uncertainty in the results because of the sample homogenization process. Composite sampling generally provides a very good estimate of the mean. However, information for the variance of the concentration needed for the final status surveys was not provided. Composited samples are also unable to detect individual areas of elevated activity (see Section 4.2 for a discussion on sampling design). The sampling methods used in the Litaor et al. (1995) study also required different numbers of samples from completely different soil horizons. Nonetheless, the two sampling methods may be comparable for the purposes of the Litaor et al. (1995) study. However, the two methods are not considered equivalent for the purpose of determining if the soil action levels have been attained at the site.

4.1.2 Depth Distribution of Plutonium and Americium

The importance of sampling depth for the sample collection protocol was identified by Bernhardt et al. (1983) as described in Section 4.1.1. The following summary of studies at the RFETS for plutonium and americium provides insight into the depth distribution of these radionuclides.

Webb et al. (1997) conducted a study using a sampling protocol developed at Colorado State University, with additional surface samples collected using the CDPHE method. This study selected thirteen 100-m^2 macroplots, which were spaced at exponentially increasing distances from the 903 Pad along each of four transects. Samples were replicated at four randomly located, 1-m^2 microplots within each macroplot. The general sampling procedure was to (1) clip the standing vegetation at ground level inside a 1250 cm^2 frame, then scrape ~3 mm of surface soil using the CDPHE scoop and template method and (2) clip the standing vegetation at ground level inside a 625 cm^2 frame, then excavate a 25 cm long × 10 cm wide area of soil in 3-cm layers to a

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depth of 21 cm using a trench technique developed earlier in Little (1976) and later modified in Webb (1992).

Webb et al. (1997) investigated the depth distribution of ^{239}Pu in the soil at Rocky Flats (Figure 4-1). To develop the generalized depth distribution for the study area, ^{239}Pu concentrations in each layer (including the 0–3-mm scrape) were normalized to the 0–3-cm concentration for each soil profile. The medians of the normalized values at each depth were then fit to mathematical functions by trial and error, but the final regression parameters were determined with commercial fitting software. In the final regression, a total of 643 individual ^{239}Pu values were used to derive the following depth relationship:

$$\left[^{239}Pu\right]_d = \left[^{239}Pu\right]_{0-3cm} \times \left[1-\left(1-1.41e^{-0.71d}-0.16e^{-0.19d}\right)^4\right] \qquad (4\text{-}1)$$

where

$[^{239}Pu]_d$ = ^{239}Pu concentration at depth d (cm)

$[^{239}Pu]_{0\text{-}3cm}$ = ^{239}Pu concentration (pCi g^{-1}) in 0–3 cm soil increment.

Webb et al. (1997) noted that the relatively uniform concentration of plutonium over the first 2 cm is suggestive of some ongoing natural mixing process.

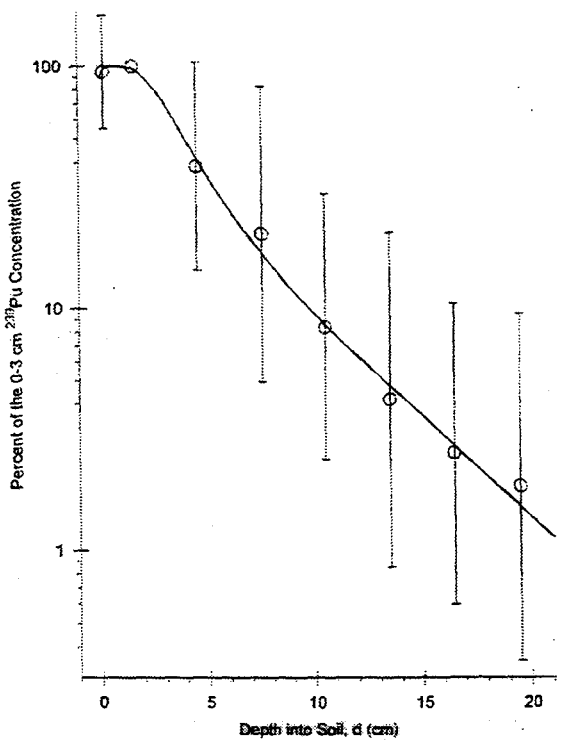

Figure 4-1. The ^{239}Pu concentrations in soil at all depths (Webb et al. 1997).

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Shierman (1994) provides information on the americium and plutonium concentration soil profiles at the Rocky Flats site. Shierman found that the concentrations of americium and plutonium decreased exponentially as a function of depth for all the locations sampled. Approximately 90% of the total inventories of the two radionuclides resided in the top 9 cm of soil, with approximately 50% of the total inventory for the two contaminants residing in the top 3 cm of soil. No difference was observed in ^{241}Am and ^{239}Pu movement vertically in the soil column.

Litaor (1999) examined the plutonium contamination in soils in open space and residential areas near Rocky Flats. Litaor included an analysis of the vertical distribution of plutonium activity from 11 soil pits collected outside RFP. The study indicated that the top layer (0–3 cm) was the most contaminated layer, with over 96% of ^{239,240}Pu activity accounted for in the top 12 cm of the soil. Below the 12-cm depth, plutonium activity decreased to background levels (see Section 4.4 for a discussion on background soil concentrations).

The studies discussed above indicate that the plutonium and americium appear to behave similarly in the soil. Each radionuclide is expected to be bound within the upper region of the soil profile, with an exponential decrease in activity with depth.

4.1.3 Depth Distribution of the Uranium Isotopes

Information on the depth distribution of uranium appears to be less extensive than that for plutonium and americium. *RAC* did not find the activity distribution of uranium with depth in the soil during a literature search. However, Litaor (1995) conducted a study of the spatial distribution of uranium isotopes in soils at the RFETS. The goal of the Litaor (1995) study was to provide information on the distribution of uranium isotopes in soils east of the RFETS. A spatial analysis of three uranium isotopes was conducted to determine the concentration and distribution pattern of uranium contamination. Geostatistical techniques were used to model the spatial dependency and construct isopleth maps showing uranium isotope distributions.

The Litaor (1995) sampling protocol required 25 equally spaced subsamples to be composited within 4.05 or 1.01-ha plots for uranium isotopes analysis. The soil at each individual location was sampled with a CDPHE sampler. Eighty-four 4.05-ha plots and thirty-four 1.01-ha plots were sampled for a total of 118 plots.

Litaor (1995) identified ^{234}U activity in soils around the RFETS that ranged from 25.9 to 92.8 Bq kg^{-1} (0.7 to 2.5 pCi g^{-1}), with a median activity of 44.4 Bq kg^{-1} (1.2 pCi g^{-1}). A spatial structure was not observed with the ^{234}U data. Litaor (1995) indicated that the lack of spatial structure suggested that ^{234}U was randomly distributed in the soil environment east of the RFETS. The randomness was also interpreted to reflect inherent irregular variation of ^{234}U dispersion in the soil that could not be predicted by the sampling method. As an alternative interpretation, Litaor (1995) also indicated that it may represent variability between sampling plots at distances less than that used in the study or samples collected from different populations (natural versus impacted because of Rocky Flats activity). On the basis of the available information, Litaor (1995) concluded that the contribution of Rocky Flats to the activity of ^{234}U in the soils was negligible.

Litaor (1995) identified ^{235}U activity in soils around the RFETS that ranged from 0.1 to 25.1 Bq kg^{-1} (0.003 to 0.68 pCi g^{-1}), with a median activity of 1.8 Bq kg^{-1} (0.05 pCi g^{-1}). The

resulting ^{235}U geostatistical contour map did not indicate the same clear west-east pattern of dispersion as those identified for $^{239+240}Pu$ and ^{241}Am (Litaor et al. 1995). The activity of ^{235}U was localized, approximately 500 m east of the industrial section of the RFETS. No clear relationship between the various sources of ^{235}U contamination considered in the study (i.e., burial trenches and the 903 Pad) and the spatial pattern of the ^{235}U isopleths was found.

Litaor (1995) states that the two soil plots with the highest ^{235}U activity probably resulted from surface flow and interflow from the east spray field. It was hypothesized that the small amounts of ^{235}U in pond water were reconstituted on the soil surface of these two plots through the continuous irrigation and subsequent evapotranspiration.

Litaor (1995) identifies ^{238}U activity in soils around the RFETS that ranged from 30.7 to 286 Bq kg^{-1} (0.83 to 7.7 pCi g^{-1}), with a median activity of 44.4 Bq kg^{-1} (1.2 pCi g^{-1}). The ^{238}U geostatistical contour map showed a clear west-east dispersion pattern like those of $^{239+240}Pu$ and ^{241}Am (Litaor et al. 1995). The highest observed activities of ^{238}U were found around the 903 Pad; however, these values did not extend beyond the immediate vicinity of that site.

Litaor (1995) notes that the complete lack of similarity in the spatial distribution across the soilscape near the 903 Pad could be explained by fundamental differences in solubility characteristics of plutonium and uranium that, in turn, affects their mode of dispersion in the environment. Plutonium is largely insoluble in the soil environment at RFETS; hence, upon removal of the drums during the cleanup operations, the impacted area became susceptible to wind and surface erosion. Consequently, plutonium particles entrapped in the fine fraction of the topsoil were airlifted by winds and subsequently deposited across the soilscape east of the 903 Pad. In contrast, under the pH and alkalinity conditions of the soil adjacent to the 903 Pad, uranium(VI) may form complex ions with carbonates and migrate downward in the soil column. Litaor (1995) notes that Seed et al. (1971) identified four hot spots below the asphalt cap that had been placed over the entire area of the 903 Pad. They retrieved >31 kg of uranium below this cap. Most of the uranium was concentrated in the B soil horizon located 15 to 40 cm below the original ground level. Increased clay content with depth decreased the hydraulic conductivity in the soils. These flow conditions facilitated the transport of uranium through the surface horizons but greatly restricted the transport to greater depths (i.e., >1 m). Litaor (1995) notes that the mobility of uranium with depth was probably further restricted because of sorption of uranium by the sesquioxides and $CaCO_3$ minerals. Because of the solubility and migratory behavior of uranium in the soil system, little uranium was entrapped in the fine particles of the topsoil. Hence, wind-dispersal mechanisms did not influence the spatial distribution of uranium isotopes across the soilscape east of the 903 Pad (Litaor 1995).

The uranium study conducted by Litaor (1995) indicates that isotopes of uranium do not behave similar to plutonium and americium in the soil environment. At the present time, the development of a soil sample collection protocol must rely on the Litaor (1995) uranium study and the assumption that the uranium was leached near the source into the soil profile and not transported by resuspension. Thus, uranium is assumed to be locally concentrated near the original source areas and present throughout the surface soil profile. *RAC* recommends that additional studies be performed similar to the Webb (1997) study on plutonium and the Shierman (1994) study for americium to determine the depth profile of the uranium contamination in the RFETS soil environment. The study by Litaor (1995) relied on samples collected from the upper 0.6 cm of the soil profile using the CDPHE sampling method and limited pit samples collected to

a depth of 9 cm. If the uranium is mobile in the soil environment, most of the uranium activity may already reside in the soil profile below that modeled for the soil action levels (i.e., 20 cm).

4.1.4 Conceptual Model Considerations

The conceptual model and assumptions for developing soil action levels are provided in Task 3, "Input and Assumptions." Two parameters used in the RESRAD model for calculating the soil action levels relate to considering sampling depth. These parameters are the depth of the soil mixing layer and the thickness of the contaminated zone.

The depth of the soil mixing layer is the depth of surface soil available for resuspension. This value is used to calculate the depth factor, or the fraction of total resuspendible soil that is contaminated. The use of this parameter in RESRAD to calculate the depth factor requires that it represent the depth over which contamination is uniformly distributed in the resuspendible layer. In the previous soil action level calculations (DOE/EPA/CDPHE 1996), the values for soil mixing layer and thickness of the contaminated zone are equal, which is not consistent with the definition of either term. *RAC* selected a value of 0.03 m to maintain consistency with the definition. This value has been used in the literature to define the surface or resuspendible soils and is the value defined by Webb et al. (1997) as representative of surface soils at Rocky Flats. We note here that the resuspension model in RESRAD was bypassed in our calculations. Instead, we used a site-specific model that accounted for the spatial distribution of plutonium in soil and annual average meteorological conditions at the site. The model was calibrated to measurements of plutonium isotopes in air at samplers located in the buffer zone and along the perimeter of the site.

The thickness of the contaminated zone in RESRAD represents the vertical distance over which radionuclide contamination levels are clearly above background. The research presented in Webb et al. (1997) indicates that plutonium contamination is distributed over the top 20 cm of soil, with very little movement of that soil within the column over the past 20 years. For this reason, *RAC* used a deterministic value of 0.2 m (20 cm) in developing the soil action levels.

4.1.5 Dominant Exposure Pathways

Soil action levels for plutonium and uranium isotopes that are reported in the Task 5 report (Killough et al. 1999) vary depending on the scenario assumed. The most limiting scenarios (those with the lowest soil action level) were the *RAC*-designated scenarios of the rancher, child of the rancher, and infant of the rancher. Soil action levels for ^{239}Pu were from 80 to 110 pCi g^{-1} for these scenarios assuming a 15 mrem dose limit and when the current vegetation was present. Under a special case where a fire removes most of the vegetative cover, soil action levels for ^{239}Pu dropped to between 10 to 25 pCi g^{-1} mainly because resuspension was substantially higher for bare soil compared to vegetated soil. Consequently, inhalation was the dominant pathway for the fire case, while the soil ingestion was the dominant pathway for the no-fire case.

The dominant exposure pathway is an important consideration for the soil collection protocol. Because surface exposure pathways such as soil ingestion and inhalation of resuspended soil dominate the dose, the upper 3 cm of soil should be sampled. While the RESRAD model provides soil action levels for the entire modeled contaminated zone (i.e., 20 cm), only the surface soil concentrations (0–3 cm) influence the dose. If the soil were sampled

over the entire contaminated zone of 20 cm, then dilution of the upper 3 cm soil profile may provide a soil concentration that is less than the action level for radionuclides despite the fact that the 0–3 cm concentration may exceed the action level. Therefore, based on the dominant exposure pathways being soil ingestion and inhalation of resuspended soil as reported in the Task 5 report (Killough et al. 1999), soil sampling should include the 0–3 cm layer.

4.1.6 Sample Compositing

The CDPHE and Rocky Flats soil sampling methods used at Rocky Flats are based on the concept of composite sampling from a systematic grid. Composite sampling consists of collecting several grab samples, from equally spaced intervals, that are thoroughly mixed into one composite sample. Then, either the entire composite is measured or one or more random subsamples from the composite are withdrawn and measured. If the mixing process is thorough, a physical averaging takes place so the subsamples represent the average concentration of the original grab samples.

Compositing is useful if (a) the cost of analyzing individual grab samples for contaminants is high, (b) the mixing process is thorough, (c) information on the variability or extreme concentrations for grab samples is not needed, and (d) the total amount of pollutant present in the composite is equal to or greater than any single grab sample making up the composite. Therefore, if the entire composite or large subsamples are analyzed, the pollutant may be more easily detected (Gilbert 1987). A major question about compositing soil samples is whether or not they can be adequately mixed. The basic idea is that the composite sample will provide an accurate average value for the individual samples used to make up the composite. If the entire composite is used for analysis, there should be no problem with the concept. For many transuranic analyses, however, only a relatively small mass is used; therefore, a composite itself may be subsampled (aliquoted) at the chemical-analysis stage. Whether compositing is worthwhile, then, depends on how well the sample is, or can be, mixed. The hot-particle problem in plutonium analysis, suggests that compositing may not be very effective (Eberhart and Gilbert 1980).

Composite sampling presents a problem for the radionuclide soil action level soil sampling protocol. Composite samples do not indicate the variance or information on extreme sample values. Therefore, small areas of elevated contaminants would not be identified by this method. Additionally, composite samples only provide an estimate of the mean soil concentration; therefore, the data do not allow a comparison to other percentiles of a distribution. For example, the mean concentration for each radionuclide may be within the action levels; however, no information is available to determine the fraction of the contaminant distribution that is above the action levels and potential hot spots are not identified.

4.1.7 Recommendations

- The recommended sample collection protocol for use in the soil action level project involves using profile sampling. The sample depth protocol recommendation assumes that soil ingestion and inhalation of resuspended contaminated soil are the dominant exposure pathways for the RFETS. The use of a profile sampling methods allows for assessing the surface soil layer for comparison to the radionuclide soil action levels for these dominant exposure pathways. In addition, information from all soil profile layers

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may be combined to provide average soil concentrations if additional pathways are determined to the important. The profile method also provides valuable information that may apply to future actinide migration studies. Using the profile sampling method in areas that have been remediated also provides information for evaluating the effectiveness of the remedial process and could be used to evaluate surrogate radionuclide ratios as described in Section 4.8.

The profile sampling method provides valuable information to determine if the sampling method is required for other remediated sites. For example, the profile data from a particular remedial area may indicate that the remediation process results in a mixed residual contamination zone in the soil so that the profile method is no longer required. In this instance, sampling for remediated areas may revert to sampling the entire upper 20 cm of soil.

- *RAC* recommends that profile sampling be conducted in soil depth intervals of 0–3-cm to be consistent with the resuspension model parameters used to develop the soil action levels.

- *RAC* recommends that soil samples not be composited for the soil action levels study. Compositing soil samples eliminates information necessary to determine if small areas of elevated contamination are present in the survey unit. Therefore, the individual soil samples should be analyzed for their radionuclide contaminants.

4.2 Sampling Design Selection

Several sampling designs are available that could be applied to the sampling protocol. These methods include random sampling, stratified random sampling, and systematic sampling using either composited samples or individual samples. These methods are discussed in detail in standard statistical texts, such as Gilbert (1987), and in EPA guidance (EPA 1991). The aim of sampling is usually to characterize a defined area, to identify unsuspected hot spots, and to demonstrate, as far as practicable, that those parts of the site believed to be less than the soil action levels are in fact less than the action levels. Sampling designs are summarized in the following sections.

4.2.1 Random Sampling

Simple random sampling is the arbitrary collection of samples within defined boundaries of the survey unit. Random sample locations are chosen using a random selection process. The arbitrary selection of sampling points requires each sampling point to be selected independent of the location of all other points, and results in all locations within the area of concern have an equal chance of being selected. Randomization is necessary to make probability or confidence statements about the sampling results. Random sampling tends to produce uneven sampling, such that large areas may not contain a sampling point. Thus, random sampling does not ensure adequate coverage of the unit to be sampled and also is not efficient in identifying hot spot areas. Figure 4-2 provides an example of a simple random sampling pattern.

contamination is expected to be far below the soil action levels, larger values of β may be appropriate, especially if final status survey sampling costs are a concern.

4.3.7 Recommendations

- As the previous discussion indicates, selecting the statistical method to analyze the data in comparison to the soil action levels involves many decisions. *RAC* recommends that the DQO planning team include a statistician familiar with the RFETS site and environmental statistical designs.

- *RAC* recommends that the arithmetic mean and its associated uncertainty at the upper 95% confidence interval be used for comparison to the soil action levels.

- *RAC* agrees with the findings of SAB (1997) that the Wilcoxon Rank Sum and Sign tests recommended in NRC (1997) are tests of the median. The MARSSIM (NRC 1997) non-parametric statistical tests should not be used for the soil action level study because these test compare the median value of the sample distribution to the soil action levels. When the distribution is not symmetrical (i.e., skewed) the median will not be equal to the mean.

- *RAC* does not recommend a specific statistical test for use in comparing the soil sample data to the soil action levels. *RAC* believes that the DQO process is the best mechanism for this selection process. Initial sampling results, or those obtained during remediation, should be evaluated to determine the potential population distribution and attributes to ensure that the most appropriate statistical techniques are employed. It is difficult to predict the affect that remediation will have in terms of changes in the present contamination attributes, including depth distribution. Various remediation strategies result in different changes in the contamination, including mixing with depth, removal of the higher contamination areas, and typically a reduction in the variance of the contamination distribution.

- *RAC* cautions that spatial correlations could impact the accuracy of the estimated mean using a systematic grid with a random starting point. Methods are available for improving the estimate of the mean using a systematic sample, and the reader is referred to Gilbert (1987) for a discussion of these methods.

4.4 Radionuclide Background Soil Concentrations

Background concentrations for the radionuclides considered in the soil action levels must be evaluated to determine their significance. As discussed in Section 4.3.2, if the background radionuclide soil concentrations are less than 10% of the soil action levels and the soil sample data distribution is symmetric (i.e., not skewed), then the non-parametric bootstrapping can be used to test the upper 95% confidence interval of the mean from the sample distribution against the soil action levels. Appendix C of the Task 5 report (Killough et al. 1999) contains a detailed discussion on background plutonium in the environment. These data are summarized in the

following section along with background data for other radionuclides. The following discussion is provided to the RSALOP to develop an understanding of background soil concentrations of important radionuclides in the Rocky Flats environment.

4.4.1 Background Studies

Background concentrations of plutonium isotopes are a result of weapons testing fallout and burnup of a Transit Navigational Satellite containing a ^{238}Pu power generator. Background studies in surface soil (Table 4-2) indicate that concentrations of ^{239,240}Pu in soil range from about 0.7 to 2.1 Bq kg^{-1} along the Front Range of Colorado. Krey (1976) calculated the global-fallout plutonium in the Denver area using ^{240}Pu:^{239}Pu ratios. The CDPHE has identified several remote locations thought to represent background levels of $^{239+240}$Pu. The $^{239+240}$Pu activity for soils collected from these areas in 1989 using the Rocky Flats sampler ranged from 0.51 to 2.84 Bq kg^{-1} (0.014 to 0.077 pCi g^{-1}), with a mean of 1.23 Bq kg^{-1} (0.033 pCi g^{-1}). Purtymun et al. (1990) studied the impact of global-fallout plutonium on remote areas in Southern Colorado and Northern New Mexico. They showed that $^{239+240}$Pu activity collected from soils ranged from 0.04 to 2.99 Bq kg^{-1} (0.001 to 0.08 pCi g^{-1}), with a mean of 1.13 Bq kg^{-1} (0.031 pCi g^{-1}). Litaor et al. (1995) also presented background soil concentrations for the Colorado Front Range and the Eastern Plains referred to as the Whiting-1994 and EG&G-1989 data sets. These reports were indicated to be unpublished by Litaor et al. (1995) and are reproduced in Table 4-2, as provided in the published paper by Litaor et al. (1995). Webb (1996) also estimated background ^{239,240}Pu in soil along the Front Range using 10 sampling sites and three sampling depths: 0–0.3 cm, 0–3 cm, and 0–21 cm. Plutonium-239,240 concentrations measured by Webb ranged from 1.2 to 3.3 Bq kg^{-1}.

Table 4-2. Background Soil Concentrations of $^{239+240}$Pu in Soils of Colorado in Bq kg^{-1} (pCi g^{-1})

Statistics	Whiting-1994[a]	EG&G-1989[a]	Krey (1976)	Webb (1996)[c]	Purtymun et al. (1990)
No. of samples	50	9	11	10	5
Mean	1.4 (0.038)	1.23 (0.033)	0.66 (0.018)	2.14 (0.058)	1.13 (0.031)
Standard deviation	0.5 (0.014)	0.73 (0.020)	0.14 (0.0038)	0.76 (0.021)	1.06 (0.029)
Range	0.62–2.66 (0.017–0.072)	0.51–2.84 (0.014–0.077)	0.40–0.92 (0.01–0.02)	1.2–3.3 (0.032–0.089)	0.04–2.99 (0.001–0.08)

[a] As reported in Litaor et al. (1995).
[b] Unpublished data. Rocky Flats, Golden, CO 80402-0464.
[c] 0–3 cm layer.

Background concentrations of ^{241}Am are also a result of weapons testing fallout. Hulse et al. (1999) evaluated background activity concentrations of ^{241}Am in 26 soil samples from depths of 0 to 3 cm (Table 4-3). The resulting data were approximately lognormally distributed with a mean of 1.3 Bq kg^{-1} (0.035 pCi g^{-1}) and 95% sign confidence interval of 1.1 to 1.6 Bq kg^{-1} (0.03 to 0.04 pCi g^{-1}). Depositions of ^{241}Am in 0 to 21 cm soil columns at 20 background locations

were also approximately lognormally distributed with a median of 0.20 Bq kg^{-1} (0.005 pCi g^{-1}) and 95% sign confidence interval of 0.15 to 0.30 Bq kg^{-1} (0.004 to 0.008 pCi g^{-1}).

DOE (1995) reported that ^{241}Am concentrations in 50 soil samples from depths of 0 to 5 cm along Colorado's Front Range ranged from 0.04 to 0.9 Bq kg^{-1} (0.001 to 0.024 pCi g^{-1}), and that concentrations in similar samples from Rock Creek, north-northeast of Rocky Flats, ranged from 0.04 to 1.3 Bq kg^{-1} (0.001 to 0.035 pCi g^{-1}).

Table 4-3. Median and 95% Sign Confidence Intervals for Concentrations and Depositions of ^{241}Am in Soil from 20 Regional Background Locations along Colorado's Front Range (Hulse et al. 1999)

Measurement	Median	95% confidence interval.
0 to 3 mm soil fraction, Bq kg^{-1} (pC g^{-1})	1.1 (0.030)	0.8–1.7 (0.02–0.046)
0 to 3 cm soil sample, Bq kg^{-1} (pCi g^{-1})	1.3 (0.035)	1.1–1.6 (0.030–0.043)
21 cm depositions, kBq m^{-2}	0.20	0.15–0.30

Concentrations of uranium isotopes are mostly from natural sources and vary depending on local geologic and geochemical conditions. Litaor (1995) presented data compiled on the background soil concentrations of the uranium isotopes. Table 4-4 presents the background data compiled by Litaor (1995) for the United States average, Colorado, and Rock Creek near Rocky Flats.

Table 4-4. Background Activities of Uranium Isotopes in Soil (Litaor 1995)

Uranium isotopes	Number of samples	Range Bq kg^{-1} (pCi g^{-1})	Arithmetic mean and standard deviation
U.S average[a]			
Uranium-238	355	4.4–140 (0.12–3.8)	37.0 ± 30.7 (1.0 ± 0.83)
Colorado[a]			
Uranium 238	32	17.3–111 (0.47–3.0)	44.4 ± 33.6 (1.2 ± 0.91)
Rock Creek[b]			
Uranium-238	21	29.6–56 (0.8–1.5)	41.6 ± 7.3 (1.1 ± 2.0)
Uranium-235	21	0.4–5 (0.011–0.124)	2.0 ± 1.3 (0.054 ± 0.035)
Uranium-234	21	28.5–54 (0.77–1.4)	41.9 ± 6.4 (1.13 ± 0.17)

[a] Statistics were taken from Myrick et al. (1983).
[b] Statistics compiled from a Rock Creek study west of the Rocky Flats Plant considered unaffected by Rocky Flats Plant activity.

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4.4.2 Recommendations

- These radionuclide background concentrations indicate that background could be excluded from the statistical analyses. The radionuclide background concentrations are near or below 1 pCi g^{-1}. This would indicate that soil action levels for each radionuclide at 10 pCi g^{-1} or higher could be statistically evaluated without consideration of background. However, it must be noted that DOE and the site contractor must determine whether they can accept the potential consequences of additional remediation by not considering background radionuclide concentrations in the statistical analyses.

4.5 Determining the Required Number of Samples

There are several equations available for calculating the number of samples required to obtain a specified level of confidence in the estimated value of the mean. These equations range from simple to more complex formulations. Equations are presented in the MARSSIM (NRC 1997) guidance for determining the number of required samples based on the Wilcoxon Rank Sum and Sign tests, and the reader is referred to this document for further information on these methods. Additional approaches to determine the required number of soil samples are provided in this section.

4.5.1 Methods

SAB (1997) presents an equation for the number of required samples for ensuring that sampling of a percentile, ω of the distribution with some probability P (e.g., P = 0.95), the relevant equation for the required sample size N is

$$N = \frac{Ln(1-P)}{Ln(\omega)} \quad . \tag{4-4}$$

The result of Equation (4-4) with P and $\omega = 0.95$ is 59 if we round up to the nearest integer. Therefore, SAB (1997) states that a sample size of about 60 will nearly always be sufficient to characterize a survey unit. However, SAB (1997) does not provide a reference for this particular sample size equation, so further investigation into its validity was not possible.

EPA (1989) presents a method for estimating the required number of samples for a systematic sampling design where the statistical parameter of interest is the mean. The sampling precision of an estimated mean from a systematic sample depends on the pattern of contamination at the site and how the systematic sample is constructed. However, EPA (1989) notes that the standard error of a mean based on a systematic sample will usually be comparable to or less than the standard error of a mean based on a random sample of the same size. Therefore, using the sample size formulas for a random sample when the sample was collected systematically will generally error on the side of conservatism. The following equation is presented by EPA (1989):

$$N = \sigma^2 \left\{ \frac{z_{1-\beta} + z_{1-\alpha}}{SAL - \mu_1} \right\}^2 \tag{4-5}$$

where

N = number of soil samples

σ^2 = standard deviation

$z_{1-\beta}$ = critical value for the normal distribution with probability 1-β

$z_{1-\alpha}$ = critical value for the normal distribution with probability 1-α

SAL = soil action level

μ_1 = the value of μ (i.e., the "true" but unknown mean contaminant concentration across the sample area, or population mean) under the alternative hypothesis for which a specified false negative rate is to be controlled ($\mu_1 < \mu$).

White and Hakonson (1979) investigated the statistical aspects of the use of the coefficient of variation, *c*, in the design of environmental plutonium studies. The number of observations, *n*, required for acceptable results is dependent on (1) the desired precision of the estimate (confidence interval length) or the power of the test to be obtained and (2) the variance of the data. Deming (1950) discusses how *c* can be used to determine the necessary sample size, *n*, to estimate a mean, $\overline{X}$ with some standard error γ, when the standard error is expressed as a percent of the mean. Deming denotes γ as the coefficient of variation of the estimate, that is

$$\gamma = \frac{SE(\overline{X})}{\overline{X}} \quad . \tag{4-6}$$

Without any knowledge of the mean, the sample size required to estimate the mean with standard error γ is

$$n = \left(\frac{c}{\gamma} \right)^2 \quad . \tag{4-7}$$

From this equation, we can see that *n* increases as the square of *c*, or that as *c* doubles, *n* quadruples. White and Hakonson (1979) evaluated the variability of plutonium concentrations in soil and reported coefficients of variation for Microplot 1 and 2 from the Little (1976) study of 2.7 and 8.4, respectively.

4.5.2 Variability of the Radionuclide Concentrations

In general, most equations for the determination of sample size require an initial estimate of the variability for the soil measurements within the survey unit. Therefore, an estimate of the standard deviation, σ, variance, σ^2, or coefficient of variation, $c = \sigma/\mu$, of the contamination distribution must be obtained. Unfortunately, the standard deviation is usually unknown and must be estimated for the purpose of determining the sample size. In practice, the estimate of σ is

either obtained from past sampling data or by conducting a small preliminary investigation in the survey unit of interest.

4.6 Classification and Identification of Survey Units

All areas of the RFETS site will not have the same potential for contamination and, therefore, will not need the same level of investigation to achieve the soil action levels. The final verification surveys will be more efficient if the surveys are designed so that areas with a higher potential for contamination receive a higher degree of investigation as recommended in NRC (1997).

The MARSSIM (NRC 1997) provides classifications for areas based upon their potential for contamination exceeding the respective soil action levels. Contamination areas must be classified to determine the appropriate survey unit size. Survey unit sizes are discussed in the Section 4.6.4. The MARSSIM classifications are described in Sections 4.6.1 through 4.6.3.

4.6.1 Class 1 Area

Class 1 areas have, or had before remediation, a potential for radioactive contamination (based on site operating history) or known contamination (based on previous radiological surveys). Areas containing contamination in excess of the action levels before remediation are classified as Class 1 areas. Examples of Class 1 areas include (a) site areas previously subjected to remedial actions, (b) locations where leaks or spills are known to have occurred, (c) former burial or disposal sites, and (d) waste storage areas.

4.6.2 Class 2 Area

Class 2 areas have, or had before remediation, a potential for radioactive contamination or known contamination but are not expected to exceed the action levels. To justify changing an area's classification from Class 1 to Class 2, the existing data from scoping or characterization surveys should provide a high degree of confidence that no individual measurement would exceed the action levels. Examples of areas that might be classified as Class 2 for the final status surveys include (a) potentially contaminated transport routes, (b) areas downwind from stack release points, and (c) areas on the perimeter of former contamination control areas.

4.6.3 Class 3 Area

Class 3 areas are any impacted areas that are not expected to contain any residual radioactivity or are expected to contain levels of residual radioactivity at a small fraction of the action levels based on site operating history and previous radiological surveys. Examples of areas that might be classified as Class 3 include buffer zones around Class 1 or Class 2 areas and areas with very low potential for residual contamination but insufficient information to justify a nonimpacted classification.

4.6.4 Identification of Survey Units

A survey unit is a physical area consisting of land areas of specified size and shape for which a separate decision will be made as to whether that area exceeds the soil action levels. The survey unit is the primary entity for demonstrating compliance with the release criterion (NRC 1997).

To facilitate survey design and ensure the number of survey data points for a site are relatively uniformly distributed among areas of similar contamination potential, the site is divided into survey units. The survey units share a common history and/or other characteristics or are naturally distinguishable from other portions of the site. Dividing the site into survey units is critical only for the final status (verification) survey; scoping, characterization, and remedial action support surveys may be performed without dividing the site into survey units.

Based upon NRC (1997), survey units should be limited in size based on classification, exposure pathways modeling assumptions, and site-specific conditions. Table 4-5 contains the suggested areas for survey units given in the MARSSIM (NRC 1997) guidance.

Table 4-5. Survey Unit Suggested Areas (NRC 1997)

Classification	MARSSIM-suggested area
Class 1—land areas	Up to 2,000 m^2
Class 2—land areas	2,000 to 10,000 m^2
Class 3—land areas	No limit

The equations for the number of samples needed for a survey unit are typically based on the variance of the contamination within a survey unit and do not consider the size (i.e., area) of the survey unit. Therefore, the limitation on survey unit size for Class 1 and 2 areas ensures that each area is assigned an adequate number of sample points. The limitations on survey unit size provided by NRC (1997) are only suggested values. The DQO process should consider additional factors for delineating survey units, including areas of remediation as survey unit boundaries.

Several considerations are associated with defining survey units, for example:

1. Multiple survey units should be specified for the RFETS. These areas should be defined so that they are as homogeneous as possible with respect to radionuclide concentrations.

2. Survey units should also be defined according to potential contamination events. For example, areas identified by Litaor (1995) that contain uranium contamination, such as from the east spray fields (see Section 4.1.3), should be identified as separate survey units from areas that were not contaminated by uranium.

3. The RFETS will contain areas that require remediation and others areas that do not require remediation. Areas that have been remediated should not be combined with areas that have not been remediated in the same survey unit. In addition, areas that have undergone different remediation techniques, for example removal versus mixing, should also not be combined in the same survey unit.

4.6.5 Recommendations

- *RAC* recommends that the MARSSIM classification scheme and limitations on survey unit size be used for the final status survey.

4.7 Small Areas of Elevated Activity (Hot Spots)

The use of systematic grid, with a random starting point, for soil sampling provides data to determine the upper 95% confidence interval of the mean for comparison to the soil action levels. Soil sampling with a systematic sampling grid may not successfully identify small areas of elevated contamination. Instead, soil sampling on a systematic grid, along with radiation detection instrumentation, are recommended to obtain adequate assurance that small areas of elevated radioactivity (i.e., hot spots) are identified during the final status survey.

4.7.1 Definitions and Detection Methods

The RSALOP and *RAC* have been involved in discussions dealing with hot spot definitions and methods to ensure that acceptable radionuclide soil concentrations are attained at the RFETS. The RSALOP has developed the following hot spot definition:

A hot spot is the location where any sample (or combination of samples) taken when following a prescribed sampling protocol that results in a radionuclide soil concentration exceeding the soil action level.

The RSALOP has also developed the following supporting statements in regard to hot spots:

- If a hot spot is found to exist, it should be evaluated to determine if action is required.

  1. Hot spots with areas equal to or greater than 100 m^2 must be remediated if the 95% upper confidence limit of the arithmetic mean soil concentrations, calculated from sample data taken in the hot spot, exceeds the soil action levels. The arithmetic mean is calculated by simple averaging of the soil concentrations in the hot spot because the hot spot encompasses the entire averaging area of 100 m^2.

  2. Remediation is required for hot spots with areas less than 100 m^2 when the area weighted arithmetic mean soil concentrations at the 95% upper confidence interval for the hot spot, when summed with the area weighted mean (95% confidence interval) of the soil concentration in the remaining 100 m^2 area (or as an alternative, the averaging area for the residential scenario), exceeds the soil action levels. The area weighted arithmetic means are used for this case (i.e., hot spot area < 100 m^2) because the hot spot area does not encompass the entire averaging area of 100 m^2.

- It is reasonable to assume that a hot spot(s) can exist within a defined area and the dose criteria for the area will still be met.

- The RSALOP may wish to establish soil action levels that if exceeded, must be remediated *regardless* of the results from averaging discussed above. The alternative action levels will ensure that small hot spots (i.e., < 25 m^2) do not contain unacceptable radionuclide soil concentrations regardless of the results of area averaging.

Identification of a hot spot, according to the above definition, requires that DOE and their site contractor, Kaiser-Hill Company, conduct additional investigations of hot spots. This can be accomplished by additional soil sampling or in situ gamma spectroscopy techniques (see the following section on surrogate measurements). The size of the hot spot must be delineated and the average radionuclide soil concentrations determined. Averaging of the hot spot over a 100 m^2 area was considered by the RSALOP to be consistent with future residential use of the land and to be protective of children that may be exposed to these hot spot in residential backyards.

The RSALOP has conducted a workshop on instrumentation for the detection of radionuclides in the Rocky Flats environment and is aware of methods and techniques available for use in the soil action level study. At Rocky Flats, the use of in situ gamma-ray spectroscopy measurements, which employ high-energy resolution germanium gamma-ray detectors, can used to identify areas of elevated soil contamination. The use of in situ gamma-ray measurements is recommended by *RAC* for identifying hot spots that may potentially be located between the soil sample locations. It should be noted that instrumentation for radiation detection is a vast field of study and new developments are continuously being announced. Therefore, *RAC* does not want to discourage the use of more recent techniques or the development of better detection instrumentation. *RAC* encourages DOE and Kaiser-Hill Company to investigate the use of these emerging technologies and techniques for application to the final status survey during the DQO process.

Recent studies have been conducted on the identifying hot spots using in situ gamma spectroscopy techniques. Reginatto et al. (1998, 1997) developed a computer code, ISD97, to analyze data from a series of in situ gamma spectroscopy measurements on a grid. The code was designed to be used as a tool when evaluating compliance with regulations that set limits on the size and magnitude of hot spots. The code calculates the location and magnitude of potential elevated activity areas consistent with the data; for each potential elevated area, it generates a corresponding distribution of radionuclides in soil. In practical applications, if any of these potential distributions appear to exceed the soil action levels, further field work (such as soil sampling, collimated measurements, or additional measurements on a closer grid spacing) would be used to determine whether such an elevated area was present. The algorithm in the code uses a maximum entropy deconvolution of the data. In deconvolution, the data and a set of additional conditions are used to find a distribution of activity in the soil that fits the data and satisfies the given set of conditions.

4.7.2 Recommendations

- *RAC* recommends the use of in situ gamma spectroscopy measurements to identify hot spot areas that may be located between the soil sampling points located on a systematic grid.

- *RAC* recommends that hot spot locations identified by soil samples or in situ gamma spectroscopy measurement be investigated further to delineate the size of the hot spot and to determine the upper 95% confidence interval of the mean concentration for radionuclides contained in the hot spot.

- *RAC* recommends that DOE and their site contractor, Kaiser-Hill Company, investigate the use of emerging techniques such as the ISD97 computer model (Reginatto et al. 1998, 1997) for use in locating hot spots during the final status survey.

4.8 Surrogate Measurements

With multiple radionuclides in the soil at Rocky Flats, it may be possible to use surrogate measurements. A surrogate radionuclide is easily measured and implies the concentration of other radionuclides.

4.8.1 Site-specific Studies

The application of surrogates has been investigated by Shierman (1994) and Hulse et al. (1999). Shierman (1994) investigated the use of ^{241}Am as a surrogate to determine ^{239,240}Pu. The ^{241}Am concentrations obtained during the study along the C transect of the 903 Pad were used to describe the ^{239,240}Pu concentrations, and the following relationship was found:

$$^{239,240}Pu\,(pCi\ g^{-1}) = {}^{241}Am\,(pCi\ g^{-1}) \times 0.189^{-1} \quad . \qquad (4\text{-}8)$$

Shierman (1994) indicated that this relationship can be used to estimate the ^{239}Pu concentration indirectly by counting the ^{241}Am via gamma spectroscopy. This would provide a quick method to quantify ^{239}Pu without expensive and labor-intensive radiochemical techniques. However, Shierman indicates that this technique may not be useful in low-level environmental samples (less than 0.27 pCi g^{-1}) because of the difficulty in quantifying ^{241}Am using gamma spectroscopy at such levels. In these cases, radiochemical techniques would be required. The ratio of ^{241}Am to ^{239,240}Pu for the RFETS soil analyzed in the Shierman (1994) study remained constant (r = 0.9946) as depth increased, but it was highly variable. Because no relationship between the ratio and depth was found by Shierman (1994), all the samples from the C transect were pooled and a median value (the distribution of the ratio data was skewed) of 0.166 was determined. The mean and standard deviation were 0.187 ± 0.94. The ratio of ^{241}Am and ^{239,240}Pu for the sampling locations on the A transect had a median of 0.189. The mean and standard deviation of the ratio for the A transect was 0.210 ± 0.85.

Hulse et al. (1999) presented a similar relationship between ^{241}Am and ^{239,240}Pu using their data and data from splits of samples reported by Webb (1996) and Webb et al. (1994, 1997). The data indicated a strong relationship between ^{241}Am and ^{239,240}Pu in soil from depths of 0–3 cm. A log-log regression yielded

$$\left[{}^{239,240}Pu\right] = 5.5\left[{}^{241}Am\right]^{1.1} \qquad (4\text{-}9)$$

with an r-squared value of 0.87 for locations where soil concentrations exceeded decision levels. They also found that the ratio between ^{241}Am and ^{239,240}Pu in soil samples from depth of 0–3 cm, which was estimated by regression, was 0.18 for all onsite locations. The ratio for offsite locations was 0.36. The ratio between concentrations of ^{241}Am and ^{239,240}Pu in soil from depths of 0 to 3 mm for all onsite locations was 0.22. This ratio was found by Hulse et al. (1999) to be the same as that reported by Mongan et al. (1996) in airborne effluents from Rocky Flats and almost the same as the ratio (0.21) reported by Litaor and Allen (1996) in soil from depths of 0 to 6 mm at their onsite sample locations. The results were also consistent with ratios calculated by Litaor et al. (1998) for concentrations in soil from depths of 0 to 20 cm and 20 to 40 cm at locations close to the 903 Pad.

Hulse et al. (1999) also used nonlinear regression to estimate parameters for the Hill four-parameter sigmoidal model that predicts the ratio of ^{241}Am to ^{239,240}Pu in soil from depths of 0 to 3 cm (Y) at distance (X in km) from the 903 Pad. The model is given by equation 4-10.

$$Y = Y_o + (aX^b)(C^b + X^b)^{-1} \tag{4-10}$$

The parameters, Y_o, a, b, C were determined to have values of 0.21, 0.21, 5.16, and 1.16, respectively. The model predicted that the ratio (95% confidence interval) should be 0.21 (0.15 to 0.29) close to the 903 Pad and 0.41 (0.35 to 0.51) at distances of more than 50 km. The predicted ratio of 0.41 at background locations was higher than the mean ratio of 0.36 estimated directly from soil concentrations in samples from offsite locations. The predicted ratio was higher than the ratio of 0.32 obtained by adjusting the ratio estimated by Krey et al. (1976) for additions of ^{241}Am from the decay of ^{241}Pu in global fallout since 1974. It was also higher than the ratio of 0.29 ± 0.17 that Hulse et al. (1999) estimated using concentrations in background soil samples reported by DOE (1995). Litaor and Allen (1996) reported that the mean ratio between ^{241}Am and ^{239,240}Pu in soil from depths of 0 to 6 mm at offsite locations around Rocky Flats was 0.56. Hulse et al. state that these ratios should be used judiciously when predicting concentrations of ^{239,240}Pu from soil concentrations of ^{241}Am to delineate the extent of contamination from Rocky Flats.

The benefit of using the surrogate approach is the reduced cost of not having to perform wet chemistry analyses on each sample. This benefit must also consider the relative difficulty in establishing the surrogate ratio, as well as the potential consequences of unnecessary investigations that result from the error in using a conservative surrogate ratio.

NRC (1997) recommends that when the ratio is established before remediation, additional post-remediation samples should be collected to ensure that the data used to establish the ratio are still appropriate and representative of the existing site condition. If these additional post-remediation samples are not consistent with the pre-remediation data, surrogate ratios should be reestablished.

4.8.2 Recommendations

- *RAC* recommends the use of surrogate measurements in the final status survey at the RFETS. The ratio of ^{241}Am to ^{239,240}Pu would provide a mechanism for the in situ gamma spectroscopy measurements of ^{241}Am to be used to predict the soil concentrations of ^{239,240}Pu.

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- *RAC* recommends that post-remediation samples be collected to ensure that the data used to establish the ratio are still appropriate, as specified in the MARSSIM guidance (NRC 1997).

4.9 Independent Confirmatory Investigations

An independent verification survey is performed by an independent third party, contracted by the DOE, to provide data to substantiate results of the final status survey. The independent verification survey would be limited in scope to spot checking conditions at selected locations, comparing findings with those of the final status survey, and performing independent statistical evaluations of the data developed from the final status survey. This task would burden the DOE with additional costs for the soil action level project; however, *RAC* considers the benefit of such an independent confirmatory investigation to outweigh the cost. The independent confirmatory investigation would provide the public with assurances that DOE and their site contractor, Kaiser-Hill Company, have conducted the final status surveys in a technically defensible manner and that decisions to release specific survey units of the site are the correct decision.

4.9.1 Recommendations

- *RAC* recommends that the RSALOP request DOE to implement a confirmatory survey (also known as an independent verification survey) for the radionuclide soil action level project. This survey should be performed by an independent third party.

5. SOIL SAMPLING QUALITY ASSURANCE

The goal of QA/QC is to identify and implement sampling and analytical methodologies that will limit the introduction of errors into analytical data. The required QA/QC program elements are typically developed and documented in QAPjPs or in similar documents, including, but not limited to, decommissioning plans, sampling and analysis plans, and field sampling plans. Section 2 discusses the RFETS sampling program and concludes that QA program elements are documented and available for use in the final status survey soil sampling protocol. However, the DQO process must be used to define the specific elements of the QA program for use in the soil action level study.

In general, the following progression of events leads to developing a QA/QC program for soil sampling programs:

1. Statement of the study objectives
2. Evaluation of the impact of mistakes
3. Definition of the DQOs
4. Design of the soil sampling study to achieve DQOs
5. Design of the QA/QC program to confirm achievements of DQOs.

Present guidance for the development of DQOs identifies the following factors for consideration in the sampling program design (NRC 1997):

- Precision
- Accuracy
- Bias
- Completeness
- Representativeness
- Comparability.

Developing a QA program for soil sampling is beyond the scope of this report. Typically, developing a soil sampling program and associated QA/QC requirements is an iterative process that the RSALOP, DOE, and Kaiser-Hill Company will need to perform during the DQO process.

In this report, *RAC* outlines the available QA/QC guidance for soil sampling programs and suggested elements for consideration by the RSALOP. Elements of a QA/QC program for soil sampling include DQOs, documentation, chain-of-custody, laboratory requirements, data validation, and the assessment of DQIs.

5.1 Data Quality Objectives and Data Quality Indicators

This section specifically targets the QA/QC aspects (or DQOs) of the soil sampling program and the method for identifying potential errors from the point of sample collection to the final analytical result. DQOs are qualitative and quantitative statements developed by data users to specify the quality of data needed from a particular data collection activity (EPA 1987).

DQOs must address five DQIs: precision, accuracy, representativeness, completeness, and comparability. Bias is also considered by the MARSSIM methodology manual (NRC 1997). Another data characteristic, level of detection, should also be addressed because it is closely

related to the other elements. Using DQIs in the QA sampling program allows researchers to determine if the data are of necessary quality to make a particular decision.

DQIs are not all quantitative (numerical) measurements; some DQIs are subject to qualitative (relative) analysis. Of the six principal DQIs, precision and bias are quantitative measures, representativeness and comparability are qualitative, completeness is a combination of both qualitative and quantitative measures, and accuracy is a combination of precision and bias.

The number of required QC measurements is determined by the available resources and the degree to which assurance is needed that a measurement process is adequately controlled. The number of QC measurements may also be driven upward as the action level approaches an instrument's detection limit.

A widely used standard practice is to collect a set percentage, such as 5% (EPA 1987), of the samples for QA purposes. However, this method has disadvantages. Depending on the number of samples to be collected, small numbers will result in insufficient QC samples, whereas large sample numbers may require too many QC samples and waste resources. A performance-based alternative is also available (EPA 1990).

The precision of an estimate of the "true" variance for precision and bias within a survey design depends on the number of QC measurements performed to provide the estimate. The MARSSIM provides one-sided upper confidence limits for selected numbers of QC measurements assuming the results of the measurements are normally distributed (see Table 5-1). At the stated level of confidence, the true variance of the estimate of precision or bias for a specified number of QC measurements will be between zero and the multiple of the estimated variance listed in the table. For example, for five field replicate samples, you would be 90% confident that the true variance for precision falls between 0 and 3.10 times the estimated variance based on the results of the five samples.

When planning surveys, the number of each type of QC measurement can be obtained from Table 5-1. For example, if the survey objective is to estimate the variance in the bias for a specific measurement system between 0 and 2 times the estimated variance at the 95% confidence level, 15 measurements of a material with known concentration (e.g., performance evaluation samples) would be indicated.

The MARSSIM recommends that the survey objective be set so that the true variance falls between 0 and 2 times the estimated variance. The level of confidence is then determined on a survey unit-specific basis to adjust the number of each type of QC measurements to the appropriate level.

Table 5-1. Upper Confidence Limits for the True Variance as a Function of the Number of Quality Control Measurements Used to Determine the Estimated Variance[a]

Number of quality control measurements	Level of confidence (%)			
	90	95	97.5	99
2	9.49	19.49	39.21	99.50
5	3.10	6.01	6.02	9.02
10	2.05	2.54	3.08	3.91
15	1.76	2.07	2.40	2.87
20	1.61	1.84	2.08	2.42
25	1.52	1.71	1.91	2.17
30	1.46	1.62	1.78	2.01
40	1.38	1.51	1.64	1.80
50	1.33	1.44	1.61	1.68
100	1.21	1.28	1.35	1.43

[a] The variance lies between 0 and x factor of the true variance at a given confidence level. For example, if 20 QC samples are taken, we are 95% confident that the true value lies between 0 and 1.84 times the estimated variance.

The DQIs are described in the following sections.

5.1.1 Precision

Precision is a quantitative measure of agreement among replicate measurements of the same property under prescribed similar conditions (ASQC 1995). Several types of replicate analyses are available to determine the level of precision. These replicates are typically distinguished by the point in the sample collection and analysis process where the sample is divided. The types of QA samples that may be used for determining precision include

- *Collocated Samples.* Collocated samples are collected adjacent to the routine field sample to determine local variability of the radionuclide concentration. Analytical results from collocated samples can be used to assess site variation but only in the immediate sampling area. They are not recommended for assessing error (EPA 1991).

- *Field Replicates.* Field replicate samples are obtained from one location, homogenized, divided into separate containers, and treated as separate samples throughout the remaining handling and analytical process. These samples are used to assess error associated with sample heterogeneity, sample methodology, and analytical procedures. Field replicates are used when determining total error for critical samples with contamination concentrations near the action level. For statistical analysis to be valid in such a case, a minimum of eight replicate samples would be required (EPA 1991).

- *Analytical Laboratory Replicate.* An analytical laboratory replicate is a subsample of a routine sample that is homogenized, divided into separate containers, and analyzed

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using the same analytical method. It is used to determine method precision; however, because it is a nonblind sample (i.e., known to the analyst), it can only be used by the analyst as an internal control tool and not as an unbiased estimate of analytical precision (EPA 1990).

- *Laboratory Instrument Replicate.* A laboratory instrument replicate is a repeated measurement of a sample that has been prepared for counting (i.e., laboratory sample preparation and radiochemical procedures have been completed). It is used to determine precision for the instrument (repeated measurements using same instrument) and the instrument calibration (repeated measurements using different instruments, such as two different germanium detectors with multichannel analyzers). A laboratory instrument replicate is generally performed as part of the laboratory QC program and is a nonblind sample. It is typically used as an internal control tool and not as an unbiased estimate of the analytical precision.

When collocated measurements are performed, an estimate of total precision is obtained. When collocated samples are not available for laboratory analysis, a sample subdivided in the field and preserved separately can be used to assess the variability of sample handling, preservation, and storage along with the variability in the analytical process, but variability in sample acquisition is not included. When only variability in the analytical process is desired, a sample can be subdivided in the laboratory before analysis.

5.1.2 Bias

Bias is the systematic or persistent distortion of a measurement process that causes errors in one direction (ASQC 1995). Bias is determined quantitatively based on the analysis of samples with a known concentration. There are several types of samples with known concentrations:

- *Reference Material.* Reference material is a material or substance, one or more of whose property values are sufficiently homogeneous and well established to be used for the calibration of an apparatus, the assessment of a measurement method, or for assigning values to materials (ISO 1993). A certified reference material contains certified values that are accompanied by an uncertainty at a stated level of confidence. When appropriate reference materials are available (i.e., proper matrix, proper radionuclide, and proper concentration range), they are recommended for use in determining the overall bias for a measurement system.

- *Performance Evaluation Samples.* Performance evaluation samples evaluate the overall bias of an analytical laboratory and detect any error in the analytical method used. These samples are usually prepared by a third party, using a quantity of radionuclides that is known to the preparer but unknown to the laboratory and always undergoes certification analysis. Laboratory procedural error is evaluated by the percentage of the radionuclide identified in the performance evaluation sample (EPA 1991).

- *Matrix Spike Samples*. Matrix spike samples are environmental samples that are spiked in the laboratory with a known concentration of a target radionuclide to verify percent recoveries. They are primarily used to check sample matrix interferences, but they can also be used to monitor laboratory performance. However, a data set of at least three or more results is necessary to distinguish between laboratory performance and matrix interference (EPA 1991).

Several types of QA/QC samples are also used to detect bias caused by contamination, which include

- *Field Blanks*. Field blanks are samples prepared in the field using certified clean sand or soil and then submitted to the laboratory for analysis (EPA 1991). A field blank is used to evaluate contamination error associated with sampling methodology and laboratory procedures. It also provides information about contaminants that may be introduced during sample collection, storage, and transport (NRC 1997).

- *Method Blanks*. A method blank is an analytical control sample used to demonstrate that reported analytical results are not the result of laboratory contamination (ATSDR 1992). It contains distilled or deionized water and reagents and is carried through the entire analytical process (laboratory sample preparation, digestion, and analysis). The method blank is also referred to as a reagent blank. The method blank is generally used as an internal control tool by the laboratory because it is a nonblind sample (NRC 1997).

5.1.3 Accuracy

Accuracy is a measure of the closeness of an individual measurement or the average of a number of measurements to the true value (EPA 1997). Accuracy includes a combination of random error (precision) and systematic error (bias) components that result from performing measurements.

Accuracy is determined by analyzing a reference material of known concentration or by reanalyzing material to which a known concentration of contaminant has been added. To be accurate, data must be both precise and unbiased. As an example, consider a target. To be accurate, the shots at the target must land close together and, on average, at the spot where they are aimed. In other words, the shots must all land near the bull's eye. Figure 5-1 shows this analogy.

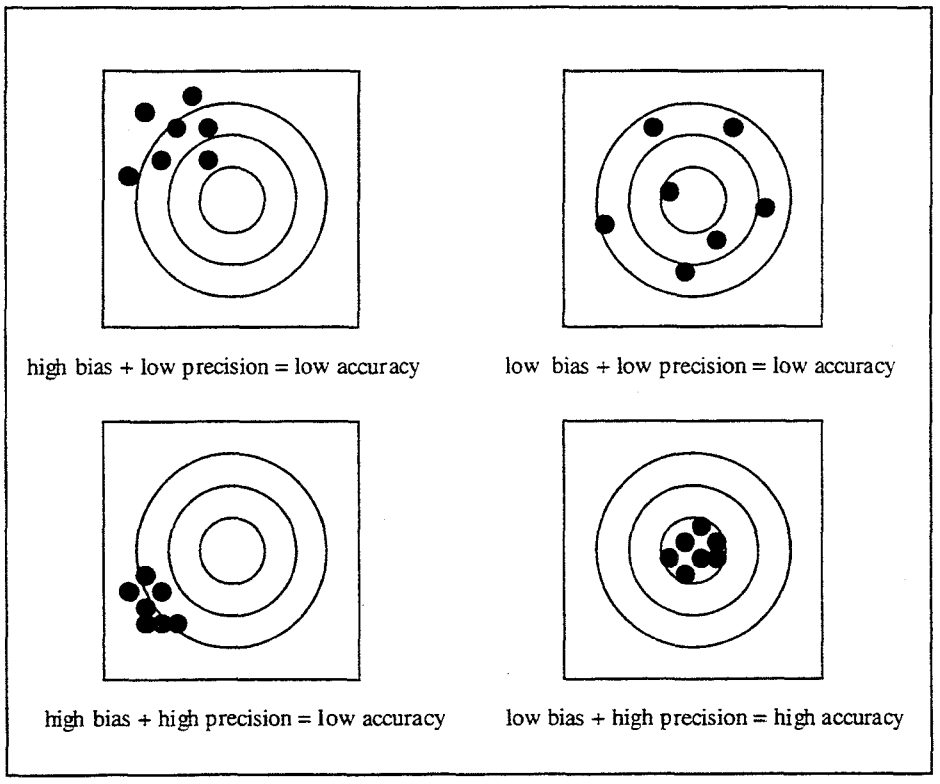

Figure 5-1. Measurement bias and random measurement uncertainty (NRC 1997).

Accuracy is usually expressed either as a percent recovery or as a percent bias. Determination of accuracy always includes the effects of variability (precision); therefore, accuracy is a combination of bias and precision.

5.1.4 Representativeness

Representativeness is a measure of the degree to which data accurately and precisely represent a characteristic of a population parameter at a sampling point (ASQC 1995). Representativeness is a qualitative term that should be evaluated to determine whether physical samples were collected in a manner that the resulting data appropriately reflect the media and contamination measured.

When soil sampling is required as part of a survey design, it is critical that the sample collection procedures consider representativeness. Sample collection procedures also need to consider the modeling basis for the radionuclide soil action levels when determining the representativeness of the samples.

Representativeness is primarily a planning concern. The solution to enhancing representativeness is in the design of the sampling plan. Analytical data quality also affects representativeness because data of low quality may be rejected for use in the analysis.

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5.1.5 Comparability

Comparability is a qualitative term that expresses the confidence that two data sets can contribute to a common analysis and interpolation. Generally, comparability is provided by using the same measurement system for all analyses of a specific radionuclide. Comparability is usually not an issue except in cases where historical data have been collected and are being compared to current analysis results or when multiple laboratories are used to provide results as part of a single sampling design (NRC 1997).

The comparability objective provides the needed control over the total measurement process to ensure that different studies can be compared. Comparability provides a basis for comparing trends over time or space, evaluating the relationship between sampling programs, or ensuring that phased sampling efforts produce data of a consistent quality.

When sampling is to occur over an extended period of time or when the investigator desires to compare several sites, it is necessary to ensure that the samples be collected in a comparable manner, from comparable fraction of the soil mass, and with comparable methods. For example, samples collected by coring should not be compared with bucket auger samples.

5.1.6 Completeness

Completeness is a measure of the amount of valid data obtained from the measurement system, expressed as a percentage of the number of valid measurements to total number of measurements collected.

Completeness for measurements is calculated by the following formula (NRC 1997):

$$\%\ completeness = \frac{(number\ of\ valid\ measurements) \times 100}{total\ number\ of\ measurements\ planned} \quad . \qquad (5\text{-}1)$$

Samples collected on a grid to locate areas of elevated activity are also a concern for completeness. If one sample analysis is not valid, the entire sample design for locating areas of elevated activity may be invalidated. If a sufficient amount of sample was originally collected, the analysis can be repeated using archived sample material.

Completeness is not intended to be a measure of representativeness, that is, it does not describe how closely the measured results reflect the actual concentration or distribution of the contaminant in the media being measured. A project could produce 100% data completeness, but the results may not be representative of the actual contaminant concentration. Alternatively, there could be only 70% data completeness (30% lost or found invalid), but because of the nature of the sample design, the results could still be representative of the target population and yield valid results.

For most final status surveys, the issue of completeness only arises when the survey unit demonstrates compliance with the release criterion and less than 100% of the measurements are determined to be acceptable. The question then becomes whether the number of measurements is sufficient to support the decision to release the survey unit.

An alternative method to ensure completeness is to take samples in addition to those determined appropriate for the sample design. The planning stages of any study must take into consideration the fact that not all samples will make it intact through the entire measurement

process. For example, sample containers will be broken, instruments will fall out of control, data will be lost, and sample tags will be lost. Many factors can lead to a sample result being invalidated. These invalid samples can be compensated for by oversampling or by using a phased sampling effort that allows areas where samples were lost to be resampled in subsequent phases. This latter approach ensures that the desired number of samples will be collected. For example, if 20% more measurements were taken in a survey design than required, then a sampling project with 80% completeness may still have sufficient power to support a decision to release the survey unit.

The design of a particular sampling effort provides a minimum number of samples needed to yield a desired level of precision for the final results. The probabilities of false positive and false negative answers are specified at the outset. Obviously, any loss from the required number of samples will impact the final results. DOE has set a completeness objective for environmental survey programs at 90% for both field sampling and laboratory analyses (DOE 1987).

5.1.7 Detection Limits

Selecting analytical methods based on detection limits is an important process. The detection limit of the method directly affects the usability of the data because results near the detection limit have increased measurement uncertainty.

5.2 Data Validation

Validation of the analytical data is the process by which the quality of the data is assessed by using the specified DQIs and QA/QC sample results. Analytical data validation, including field and laboratory data review, is defined as the systematic process, performed external from the data generator, that applies a defined set of performance-based criteria to a body of data and may result in qualification of the data. Data validation provides a level of assurance, based on a technical evaluation, that an analyte is present or absent; if present, it provides the level of uncertainty associated with the measurement. Data validation must occur before drawing a conclusion from the data.

Analytical data validation for radiochemistry includes a technical review of the laboratory data package covering the evaluation of DQI samples, the identification and quantitation of analytes, and the effect of deficiencies in QC on analytical sample data.

Although the EPA has developed numerous guidance documents relating to data validation of organic and inorganic constituents, no national standard currently covers data validation of radiochemistry concepts adequately. The need for a document of this type has been recognized by most of the DOE complex. There is reference in the MARSSIM (NRC 1997) to the development of such guidance, the *Multi-Agency Radiation Laboratory Analytical Protocols (MARLAP) Manual;* however, currently, no such document exists. Because of the lack of specific guidance, currently each DOE site has developed site-specific data validation procedures for radiochemistry data.

During data validation, the reviewer examines the data, documentation, and reports to determine if the sampling program was conducted within the limits specified by the DQO process.

5.3 Documentation

Three types of documentation are available for review during the data validation process: (1) field operation records, (2) laboratory records, and (3) data handling records (EPA 1997).

5.3.1 Field Operation Records

The information contained in these records documents the field operations and consists of the following:

- *Field Measurement Records.* Field measurement records provide documentation that the proper measurement protocol was performed during the sampling project. This documentation includes the names of the persons conducting the sampling, sample location and identification, maps and diagrams, equipment and SOP used during sampling, and unusual observations. Bound field notebooks are generally used to record raw data; however, data recording forms may also be used for this documentation.

- *Sample Tracking Records.* Sample tracking records, also referred to as chain-of-custody records, document the progression of samples as they travel from the original sampling location to the laboratory and finally to disposal.

- *General Field Procedures.* General field procedures, also referred to as SOPs, record the procedures used in the field for collecting soil samples.

5.3.2 Laboratory Records

The following list describes some of the laboratory-specific records that should be reviewed if available and appropriate:

- *Laboratory Measurement Results and Sample Data.* Laboratory measurement results and sample data contain information on the sample analysis used to verify the analytical methods that were followed. The overall number of samples, sample identification, sample measurement results, any deviations from the SOPs, time of day, and date should be included. Sample location information may also be provided.

- *Sample Management Records.* Sample management records should document sample receipt, handling, and storage and scheduling of analyses. The records will verify that sample tracking requirements were maintained; reflect any anomolies in the samples, such as receipt of damaged samples; and note proper log-in of samples into the laboratory.

- *QC Measurement Records.* QC measurement records include general QC records, such as initial demonstration of capability, instrument calibration, routine monitoring of analytical performance, and calibration verification. Project-specific information from the QC checks, such as blanks, spikes, calibration check samples, replicates, and splits, should be included in these reports to facilitate data quality analysis.

5.3.3 Data Handling Records

Data handling records document protocols used in data reduction, verification, and validation. Data reduction addresses data transformation operations, such as converting raw data into reportable quantities and units, using significant figures, and calculating measurement uncertainties. The records document procedures for handling data corrections.

5.4 Data Validation Qualifiers

Data validation begins with an assessment of the quality of analytical results and is performed by a professional with knowledge of the analytical process. Depending on the survey objectives, the level and depth of review varies. The level and depth of the data validation may be determined during the planning process and should include an examination of laboratory and method performance for the measurements and radionuclides involved. This review includes

- Evaluating data completeness
- Verifying instrument calibration
- Measuring precision using replicates or split samples
- Measuring bias using reference material or spikes
- Examining blanks for contamination
- Assessing adherence to method specifications and QC limits
- Evaluating method performance in sample matrix
- Assessing applicability and validation of analytical procedures or site-specific measurements
- Assessing external QC measurement results and QA assessments.

Following the data validation process, data are assigned validation qualifiers. The person conducting the data review assigns coded qualifiers to the data when QC requirements or other evaluation criteria are not met. An explanation of the data qualifiers should be included in the data validation report, along with a summary of the quality of the data package.

6. CONCLUSIONS

The primary goal of Task 6 is to develop recommendations for a soil sampling protocol for use at RFETS to obtain soil concentration data for comparison to the soil action levels. Sampling protocols are written descriptions of the detailed procedures to be followed in collecting, packaging, labeling, preserving, transporting, and documenting the samples.

Sampling protocols are developed using the iterative DQO process and require DOE and its contractor to evaluate several important considerations. These considerations include evaluating sampling and analytical costs in relation to available resources and accepting potential decision errors that may result in remediating sites that are judged contaminated when they are actually below the soil action levels. Conversely, developing a sampling protocol must also incorporate the concerns of the general public and other stakeholders, which are represented by the RSALOP and the soil action level study. Because of the complexity of developing sampling protocols, with the inherent need to balance the concerns of DOE and the RSALOP, developing a comprehensive sampling protocol was not considered possible. In this report, *RAC* presents recommendations for those elements of a soil sampling protocol considered essential to ensure representative soil samples are collected for comparison to the soil action levels. These recommendations are provided to the RSALOP for presentation to DOE and its contractor, Kaiser-Hill Company, for incorporation into the soil sampling protocol and procedures to be used for the soil action level study.

The sampling protocol recommendations were developed for the surface-soil sampling (i.e., 0 to 20 cm) during final status survey. In this report, *RAC* used the general principles of the MARSSIM (NRC 1997) guidance to develop recommendations for a sampling protocol. *RAC* has provided an emphasis in this report on problems identified with the MARSSIM guidance in terms of its application to the soil action levels at Rocky Flats and potential solutions.

The following is a summary of the recommendations for a soil sampling protocol in support of the final status survey. *RAC* recommends that

1. The DQO process be used to develop the soil sampling protocol for the final status survey.

2. DOE appoint representatives from the RSALOP for inclusion on the DQO planning team.

3. The RSALOP select discrete values from the soil action level distributions for each radionuclide and use these discrete values for comparison to the soil concentration data.

4. Soil samples be collected using profile sampling.

5. Profile sampling be conducted in soil depth increments of 0–3 cm to be consistent with the resuspension model parameters used to develop the soil action levels.

6. Soil samples should not be composited; rather, individual soil samples should be analyzed for radionuclide contaminants.

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7. Soil samples be collected using a systematic grid sampling design, with a random starting point.

8. A statistician familiar with the RFETS and environmental statistical designs be included on the DQO planning team.

9. The arithmetic mean of the soil concentration data and its associated uncertainty at the upper 95% confidence interval be used for comparison to the soil action levels.

10. The MARSSIM (NRC 1997) non-parametric statistical tests not be used for the soil action level study since these test compare the median value of the sample distribution to the soil action levels. When the distribution is not symmetrical (i.e., skewed), the median is not equal to the mean.

11. Parametric statistical tests, bootstrapping, or geostatistical techniques be investigated for use in comparing the 95% upper confidence limit of the mean against the soil action levels. The statistical tests should be investigated during the DQO process and chosen according to knowledge of the areas to be sampled.

12. Spatial correlations be investigated to determine their presence in the survey unit of interest and to determine if methods are required to improve the estimate of the mean based on the systematic grid sampling method.

13. The null (Ho) and alternative (Ha) hypothesis are stated as Ho: $\mu \geq$ SAL and Ha: $\mu \leq$ SAL, where μ is the mean soil concentration and SAL is the soil action level.

14. The survey units be classified according to the NRC (1997) scheme and that the size of the survey units be limited accordingly.

15. In situ gamma spectroscopy measurement be performed to identify potential hot spot locations.

16. Hot spots identified by soil samples or in situ gamma spectroscopy measurements be investigated further to delineate the size of the hot spot and to determine the upper 95% confidence interval of the mean radionuclide concentrations contained in the hot spot.

17. That hot spots greater than 100 m^2 with arithmetic mean soil concentrations at the upper 95% confidence interval that exceed the soil action levels be remediated.

18. That hot spots less than 100 m^2 be area averaged with soil concentrations in a 100 m^2 area and area weighted to determine if the upper 95% confidence interval of the mean soil concentration exceeds the soil action levels and, thus, requires remediation.

19. The use of surrogate measurements in the final status survey. The ratio of ^{241}Am to $^{239,240}Pu$ provides a mechanism for the in situ gamma spectroscopy measurement of ^{241}Am to be used to predict the soil concentrations of $^{239,240}Pu$.

20. DOE implement an independent verification survey for the radionuclide soil action level project. This survey should be performed by an independent third party.

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APPENDIX D

INTERACTION WITH ACTINIDE MIGRATION EVALUATION GROUP

ATTACHMENT D

SUMMARY OF ACTINIDE MIGRATION EVALUATION MEETINGS

The Actinide Migration Evaluation (AME) projects were established by DOE in 1996 to investigate and model $^{239,240}Pu$, ^{241}Am, and uranium transport in the site environment. Periodic technical and public meetings have been a way for Kaiser-Hill (K-H) and the various scientists who specialize in the study of actinide movement in the environment to report on the progress of their findings at Rocky Flats. The specific projects that are currently underway are

- Watershed erosion and sediment transport modeling
- Air transport modeling
- Biological impact on actinide mobility
- Pathway analysis
- Updating the data quality objectives for the AME project
- Uranium geochemical modeling
- Colloidal transport and aggregation experiments
- Sampling process waste lines for plutonium, americium, and uranium oxidation states
- Installation and sampling groundwater aseptic wells
- High Resolution Inductively-Coupled Plasma/Mass Spectrometry (ICP/MS) analysis of RFETS groundwater and surface water samples
- Soil aggregation/disaggregation studies by Colorado School of Mines (funded by U.S. Environmental Protection Agency (EPA).

A *Risk Assessment Corporation (RAC)* representative attended the AME meetings beginning with the meeting on November 19, 1998, at the Arvada City Hall. The group meets about every 6-8 weeks. As we began the project, there were four reports from K-H and Rocky Mountain Remediation Services (RMRS) that were currently available about the work:

RMRS. 1998. Loading Analysis for the Actinide Migration Studies at the Rocky Flats Environmental Technology Site. Rev. 0. RF/RMRS-98-277. UN. September. (Available surface water discharge and actinide activity data from site monitoring programs during the 1990s were compiled to compute actinide loads on a storm-specific and annual basis. The analysis was done for Woman Creek, Walnut Creek and the South Interceptor Ditch [SID] drainage basins, which is part of the Woman Creek watershed).

RMRS. 1998. Actinide Content and Aggregate Size Analyses for Surface Soil in the Walnut Creek and Woman Creek Watersheds at the Rocky Flats Environmental Technology Site. Rev. 1. RF/RMRS-98-281.UN. September. (Reports $^{239,240}Pu$, and ^{241}Am activity in surface soil sampled in Fiscal Year 1998 from the Walnut and Woman Creek watersheds. Particle size distribution of the soil and sediment samples were done at the Colorado School of Mines. Data will be used as source of actinides to streams via storm water runoff and to calibrate the Watershed Erosion Prediction Project [WEPP] model to estimate soil erosion and associated actinide transport).

RMRS. 1998. Conceptual Model for Actinide Migration Studies at the Rocky Flats Environmental Technology Site. No number on cover. October.

RMRS. 1998. Preliminary Report on Soil Erosion/Surface Water Sediment Transport Modeling for the Actinide Migration Study at the Rocky Flats Environmental Technology Site. Fiscal Year 1998. RF/RMRS-98-285.UN. November. (Provides preliminary modeling results that will be used for calibrating the soil erosion and surface water transport modeling effort for the Rocky Flats watersheds. This report describes results for the SID watershed, which drains into Pond C-2).

At the November 19, 1998 meeting, there were two major topics: (1) soil erosion and sediment transport modeling Fiscal Year 1999 work (a summary of the November 1998 report above) reported by Win Chromec and Ken Spitze, and (2) Air Modeling Fiscal Year 1999 work that will be done by Radian International reported by Martha Hyder and Arney Srackangast. Radian International is just beginning the current air modeling work and will focus on "improving estimates of airborne actinide migration and deposition in the conceptual model," (paper three above) "preparing a modeling tool to use in evaluating various emission scenarios, and providing preliminary air pathway dose estimates." They plan to do this by reviewing published studies of contaminated soils resuspension to determine resuspension mechanisms, and then to identify resuspension models to use in estimating emissions of actinides from contaminated soils into the air.

The January 21, 1999 AME meeting focused on studies of plutonium migration at the Rocky Flats site. Dr. Greg Chopin from the University of Florida described his work with the use of oxidation state actinide analogs to observe effects of geochemical processes over long time periods. He and his colleagues have studied old uranium and thorium locations around the world to find analogs for plutonium, (e.g., Th^{4+} for Pu^{4+}, and U^{4+} for Pu^{4+}). The main message is that natural analog sites provide valuable information on actinide chemistry and fate and transport; to date these studies show very little movement of plutonium over long time periods. Their studies indicate that Rocky Flats plutonium is insoluble but they emphasize that solubility studies are complex. At that same meeting, Mike Murrell and Chris Brink from Los Alamos National Laboratory (LANL) explained how they are tracing uranium migration at the Rocky Flats solar ponds using refined analytical techniques in ion counting to follow the transport of uranium and to differentiate between "Rocky Flats" uranium and "natural" uranium.

At the April 29, 1999 AME meetings, researchers described progress on collecting borehole samples from the South Interceptor Ditch, runoff samples from a buffer zone area near Walnut Creek, and water samples from Pond B-5 discharge that will be used for suspended solid fractionation experiments. Jim Ranville from the Colorado School of Mines, described his work on soil aggregation at Rocky Flats and how it might affect solubility. Mary Neu from LANL then described results of current experiments done on characterizing plutonium in samples from the 903 Area. Using powerful, state-of-the-art analytical techniques, she and her colleagues 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 provide solid proof for what many have assumed all along—that plutonium in the soil at Rocky Flats is insoluble PuO_2, and thus may not get into the groundwater.

These studies are exciting and very relevant to the current soil project because they help to characterize the chemical and physical form of plutonium at the Rocky Flats site. The AME

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research that is underway has helped to define the potentially significant pathways and we still see inhalation as the major pathway for this work. Recent work at LANL indicates that the plutonium from soil samples under the 903 Pad is insoluble PuO_2. While results from some of the AME studies indicate that this insoluble form of plutonium may not enter groundwater, we are examining a conservative calculation to address the question of whether or not the pathway can be ruled out of the current analysis. We understand the importance of groundwater and surface water pathways in the long term, and include the groundwater pathway in one of our scenarios. We do recognize, however, that our assessment of the groundwater pathway is limited by the complexity of the pathway.

On October 4, 1999, the AME meeting covered the following topics: an update of the site conceptual model by Chris Dayton; a summary of experimental results obtained to date by the Colorado School of Mines and by Texas A&M, Galveston; results of the FY99 atmospheric dispersion and deposition by Martha Hyder of Radian International; biological mobility of environmental plutonium by Ward Whicker of Colorado State University and an update on the 903 Pad remediation project by Steve Paris of RMRS. The characterization of the 903 Pad is complete and a remediation strategy is being developed.

A study by the Colorado School of Mines that is investigating the effect of environmental redox potential on the solubility of plutonium shows that under moderately reducing environments while the soil is fairly saturated, there is less plutonium found in the <0.45 μ size fraction than in oxidizing environments. Americium's association with a particular size fraction seems to independent of the redox potential.

To evaluate the biological mobility of environmental plutonium, a draft "white paper" by Dr. Kathryn Higley of Oregon Sate University and Dr. Ward Whicker of Colorado State University, titled "Biological Mobility of Environmental Plutonium," was distributed. Dr. Whicker described the results of their study and the report is available through the Citizen's Advisory Board. Plutonium is not a biologically essential nutrient, nor an analog of a biologically essential nutrient. Much of the plutonium measured in plant material comes from atmospheric deposition onto the leaf surface.

APPENDIX E

RADIONUCLIDE SOIL ACTION LEVEL OVERSIGHT PANEL

RSALOP

Radionuclide Soil Action Level Oversight Panel

Planning for Tomorrow... Radionuclide Soil Action Levels at Rocky Flats

BACKGROUND

Rocky Flats operated from 1952 - 1992 as a manufacturing facility for the production of component parts for nuclear weapons, including plutonium pits, the fissionable core of such weapons. Since cessation of production activities, the site has been involved in waste management and environmental remediation activities. One of the primary challenges facing site officials and residents from surrounding communities is determination of the amount of radionuclides, such as plutonium, that may legally remain in the soil following remediation. These levels are known as "radionuclide soil action levels" (RSALs) because remediation action is triggered when the amount of radioactive material in the soil exceeds the established levels.

Using a computer-modeling program known as RESRAD (Residual Radiation) that was developed at Argonne National Laboratory, interim radionuclide soil action levels were incorporated into the Rocky Flats Cleanup Agreement (RFCA) on October 18, 1996. The RFCA serves as the legally binding agreement for site remediation between the Department of Energy and its regulators -- the Environmental Protection Agency and the Colorado Department of Public Health and Environment. Intended to be protective of people using the site after closure, the RSALs set the upper limits for the radionuclides (primarily plutonium and americium) allowed in the soil at Rocky Flats after remediation. The RSALs are measured in picocuries (a measure of radiation) per gram of soil.

Almost immediately after the RSALs were established, members of the community became concerned. Not only were the numbers much higher than anticipated, but upon further investigation the levels were found to be higher than levels established for remediation at sites elsewhere, as shown below:

Location	Action Level for Plutonium (in picocuries per gram of soil)
Rocky Flats Environmental Technology Site	*1,429*
Nevada Test Site	*200*
Enewetak Atoll (South Pacific bomb test site)	*40*
Hanford Nuclear Reservation (Washington State)	*34*
Johnston Atoll	*15*

PUBLIC REACTION

After looking at the above numbers, it became apparent that more information was needed to better understand the criteria used for the soil action levels. As a result, several entities, including the Rocky Flats Citizens Advisory Board, the cities of Westminster and Broomfield, public interest groups, and Congressman David Skaggs called for an independent assessment of the RSALs as well as the process used to establish those levels. The end result of ensuing discussions was an unprecedented agreement by the Department of Energy to fund a community-directed, independent scientific assessment of the RSALs for Rocky Flats. To provide oversight of the study, a panel of thirteen community representatives was formed, known officially as the Radionuclide Soil Action Levels Oversight Panel (RSALOP).

After drafting and issuing a formal Request for Proposals, the Panel selected Risk Assessment Corporation (RAC) to conduct the assessment. RAC has previous experience with the Rocky Flats site studying the potential health effects caused by site emissions during production years. In addition, RAC has more than twenty years experience working on dose

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reconstruction, environmental dosimetry, chemical risk analysis and related disciplines. RAC has assembled a team of technical experts, each with a particular area of expertise necessary to complete the RSAL assessment. Work began in October 1998 and is scheduled for completion in November 1999.

PROJECT SUMMARY

Several project tasks have been defined for the contractor. The first of these is to research RSALs used at other plutonium-contaminated sites around the world. Second, RAC is tasked with exploring existing computer models that could be used to determine RSALs and recommending the one best suited for use at Rocky Flats. Third, RAC will investigate and then recommend pertinent inputs and assumptions for use in computer calculations. Fourth, with the preceding information, RAC's major task will be to conduct an independent calculation of the RSALs for Rocky Flats, which can then be compared with the originally recommended RSALs. The final two tasks involve recommending appropriate soil sampling procedures to determine the amount of contamination in soil both before and after remediation activity. Finally, RAC will analyze results from the ongoing Actinide Migration Studies to determine whether any findings from these studies are applicable to the soil action levels.

PUBLIC INVOLVEMENT

Although public involvement began when community representatives approached the Department of Energy with concerns that resulted in formation of the RSALOP, the Panel has developed a public involvement and information strategy to ensure ongoing communication. Monthly working meetings of the Panel are open to the public and are conducted the second Thursday of each month from 4-7 p.m. at the Broomfield City Center, One Descombes Dr., Broomfield, CO. In addition, three public information workshop/meetings will be scheduled during the project. With some up-front planning, a member of the Panel can provide briefings to community groups or interested parties. For a project status update, information packet, or further information on meeting times and locations, please contact either Carla Sanda, Advanced Integrated Management Services, Inc., at (303) 277-0753 or Ken Korkia, Rocky Flats Citizens Advisory Board, at (303) 420-7855.

Introducing the RSALOP Team...

Panel Co-Chairs

Mary Harlow serves as the Rocky Flats Coordinator for the City of Westminster, acts as the City liaison to other City, County, State, and Federal organizations relative to Rocky Flats activities, and keeps the City Council and staff apprised of pertinent site issues. She represents the City staff on the Rocky Flats Coalition of Local Governments Board of Directors and is currently Secretary for the Rocky Flats Citizens Advisory Board. Additionally, she serves on the Board of Directors of the Energy Communities Alliance, a national organization of local communities that are located in the shadow of Department of Energy facilities. Ms. Harlow has a BS in Health Care Management, an AAS in Environmental Technology and an AS in Water/Wastewater Technology.

Henry A. Stovall earned a BS in Industrial Education and a BS in Physics. Prior to retirement, he was an Engineering Manager with 33 years experience in the engineering arena, including environmental health and safety engineering. Mr. Stovall serves as a member of the Rocky Flats Health Advisory Panel and has been actively involved in related Rocky Flats issues. As a twenty-nine year resident of Broomfield, Colorado, Mr. Stovall has been active in various community activities, including serving on the Broomfield City Council from 1977 - 1993 and again from 1995 - present.

Panel Members

Tom Davidson received his BS in Electronic Engineering and has thirty years of engineering experience, including 16 years in the nuclear field. He is a member of the Rocky Flats Citizens Advisory Board and has participated in numerous issues related to the cleanup and future use of the site. Mr. Davidson is also an active participant in the University of Colorado at Boulder Chancellor's Community Advisory Council. He currently serves as the Mayor of Louisville, Colorado and represents his community on the Panel.

Joe Goldfield earned his BS in Chemical Engineering and achieved professional affiliation with the American Academy of Environmental Engineers as a Diplomate. His professional career with Manville Corporation spanned thirty-one years, of which 20 years was spent as manager of Environmental Engineering. He holds a Professional Engineers' license in New Jersey, Colorado, Massachusetts, and Mississippi and owns several patents on pollution control devices. Mr. Goldfield has been an active stakeholder in numerous issues at the Rocky Flats site.

Dean Heil is currently Assistant Professor of Soil Chemistry in the Department of Soil and Crop Sciences at Colorado State University. He completed his Ph.D. in Soil Science at the University of California at Berkeley. His past research includes remediation of Pb (lead) polluted soils.

Robert Kanick earned a BS in Nuclear Engineering with an emphasis in reactor/power operations. His experience as a reactor engineer and core designer has provided valuable experience ranging from the development and oversight of reactor testing and control of special nuclear materials, to the use and evaluation of computer modeling systems used for core design, safety and criticality analysis. As a member of the

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Rocky Flats Citizens Advisory Board, Mr. Kanick has been actively involved in numerous projects related to the cleanup and future use of the Rocky Flats site.

Carol E. Lyons serves as the Rocky Flats Coordinator for the City of Arvada, Colorado.

Todd Margulies earned his MS in Environmental Sciences/Geochemistry, as well as a Bachelor's in Geology. After spending more than ten years working for major environmental firms, he began work as an independent environmental consultant in 1991. He has dealt with numerous Rocky Flats issues and was a staff member of the Colorado Council on Rocky Flats. In addition, Mr. Margulies has conducted groundwater, surface water, soil, and sediment investigations at sites around the world to assess organic and inorganic contaminant transport and fate.

LeRoy Moore, Ph.D., serves as a consultant to the Rocky Mountain Peace and Justice Center in Boulder, Colorado. Author of the Citizens Guide to Rocky Flats (1992), Dr. Moore is a member of the Rocky Flats Citizens Advisory Board. Involved with the Rocky Flats issue since 1979, he has played a key role in numerous projects, including the Rocky Flats Future Site Use Working Group. He co-chairs the recently created Rocky Flats Actinide Migration Studies Technical Review Group. He is currently working with a photographer on a book which will tell the Rocky Flats story in images and words of workers who made bombs inside the facility and activists who opposed such production from the outside.

Lisa Morgan Morzel serves on the Boulder City Council, having been elected in 1995. As a council member, she represents Boulder on issues related to Rocky Flats, including the Rocky Flats Local Impacts Initiative and its successor group, the Rocky Flats Coalition of Local Governments. Dr. Morzel is a research geologist for the U.S. Geological Survey and holds a Ph.D. in Geology and Geophysics.

Niels Schonbeck is a professor in the Department of Chemistry at Metropolitan State College of Denver, Colorado, where he has taught since 1978. He has also been a visiting scientist at the National Center for Atmospheric Research in Boulder, Colorado. He earned his Ph.D. in Biological Chemistry and his BA in Chemistry. In 1987 he became involved in the health and safety issues of the Rocky Flats site and was appointed to the Rocky Flats Environmental Monitoring Council. Since then he has also served as a member of the Rocky Flats Health Advisory Panel.

Joel Selbin received his BS in Chemistry and his Ph.D. in Inorganic Chemistry. He joined the faculty at Louisiana State University (Baton Rouge) in 1957, where he served until 1991. During his 34 years at LSU, he directed the research of graduate and postdoctoral students, authored and collaborated on numerous books and articles, and obtained two U.S. patents. He has written, lectured, and debated widely on nuclear power and nuclear arms issues, as well as energy alternative issues, and has served as a technical consultant to numerous academic and governmental entities. Upon his retirement from LSU, Dr. Selbin relocated to Boulder, Colorado, where he taught for seven years at the University of Colorado in Denver. He now teaches at the University of Colorado-Boulder campus.

Ken Starr has more than 13 years' experience in the fields of civil and environmental engineering including numerous CERCLA/Superfund remediation and investigation tasks, environmental assessments, environmental, health and safety assessments, as well as health and safety training. He currently serves as the Director of the Environmental Compliance Division for the Jefferson County Department of Health and Environment. He holds an MS in Environmental Engineering, a BS in Agricultural/Civil Engineering, and is a Registered Professional Engineer for both the States of Colorado and Arizona.

Technical Support Contractor

Since its formation in 1977, Risk Assessment Corporation (RAC) has significantly contributed to the development and application of methods for estimating exposures from radionuclides and chemicals to the public and workers and quantifying the health risks and their uncertainties. RAC has also encouraged public participation in dose reconstruction studies and has developed innovations in the communication of exposure and risk information to the public. The following team of professionals, headed by RAC's president, Dr. John E. Till, has been assembled to conduct this study:

John E. Till holds a Ph.D. in Nuclear Engineering, an MS in Health Physics, and a BS in Engineering. In addition, he is a graduate of the U.S. Naval Academy and the U.S. Naval Nuclear Propulsion Program. Following his naval career, Dr. Till worked as a staff scientist at the Oak Ridge National Laboratory, and in 1977 formed Risk Assessment Corporation (RAC). Dr. Till's scientific achievements include more than 150 publications, including editing the first textbook on radiation dose analysis, Radiological Assessment, and other documents that stress new approaches to apply and simplify risk analysis. Dr. Till's current work focuses on the assessment of risks from past releases of radionuclides and chemicals.

George G. Killough earned an MA degree in mathematics from the University of Tennessee at Knoxville and did further study under a grant from the National Aeronautics and Space Administration. His more than 30 years of professional experience include university teaching (mathematics and computer science) and serving as a member of the research staff at Oak Ridge National Laboratory, where he designed and implemented mathematical models and applied them to environmental problems. He developed internal dosimetry computer software, which was subsequently the basis for the Environmental Protection Agency's RADRISK program. Under sponsorship of the National Science Foundation and the Department of Energy, he constructed models of the global cycling of carbon and tritium, which were applied to estimating levels of carbon dioxide as a greenhouse gas and to predicting exposure of the world population to releases of 14C and 3H from nuclear fuel cycles. As a consultant to Risk Assessment Corporation, he participated in a dose reconstruction project for the Fernald nuclear facility, sponsored by the Centers for Disease Control and Prevention. For the Fernald study, he devised new methods for confirming estimates of airborne uranium releases from soil measurements and modeling the atmospheric diffusion of radon decay products using radon monitoring data. Mr.

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Killough is author or co-author of many refereed publications. In his current work, he makes innovative use of probabilistic methods and Monte Carlo bootstrap sampling to assess uncertainties in model predictions and to calibrate environmental models to site-specific data.

Kathleen R. Meyer holds an MS degree in health physics and a Ph.D. in radiological health sciences. She has served as a research assistant at Oak Ridge National Laboratory where she assessed the radiation damage and subsequent recovery capabilities of normal tissue, both in cell culture and in animal models. Her career has also included independent work in radiological dose assessments, technical abstracting, and chemical and radiological risk evaluation for sites containing hazardous materials. Dr. Meyer has more than 20 years experience in the fields of biological research and teaching, radiation protection, and public communication.

Arthur S. Rood received his Bachelor's degree in geology and an MS in health physics. His experience includes environmental and laboratory measurements of uranium-related byproducts. He later joined a team of environmental scientists at the Idaho National Engineering and Environmental Laboratory and worked primarily in the field of environmental contaminant transport modeling and dose assessment. Mr. Rood has now formed his own corporation, and projects currently focus on the use of atmospheric dispersion models to study past releases of chemicals and radionuclides to support dose reconstruction studies at the Rocky Flats Plant and Savannah River Site. Mr. Rood also teaches an Environmental Modeling class for the University of Idaho.

David J. Thorne received his Bachelor's degree in geology and his MS degree in health physics. His career has included surveillance of the Fort St. Vrain Nuclear Power Station and conducting studies on concentrations of 131I in dairy milk. He has also been involved in providing air quality modeling support, preparing radiological health risk assessments, conducting studies on groundwater transport of contaminants, and development of an atmospheric dispersion modeling study as well as computer models to simulate the transport of radionuclides. Mr. Thorne is currently using his extensive experience with computer systems, computer programming languages, and contaminant transport models to test computer models, assess residual contamination in buildings, model accidental releases, develop risk-based standards, and model potential groundwater impacts.

Jill M. Weber received her undergraduate degree in 1993 in physics with minors in chemistry and mathematics and her MS degree in radiological health sciences with a specialty in health physics. Ms. Weber joined RAC in 1995 and has been involved in a number of radiological dose reconstruction projects. Her research has included model development for releases of plutonium from the 903 Pad Area at Rocky Flats, a contaminated soil area that was exposed to high winds and resulted in significant releases. She also has experience in environmental monitoring, uncertainty estimation, and public communication.

Administrative Support Contractor

Advanced Integrated Management Services, Inc. (AIMSI) is a small, minority-owned, Tennessee corporation founded in 1994 with a staff of over 100 full- and part-time professionals. Headquartered in Oak Ridge, Tennessee, with field offices in Denver, Colorado, Nashville, Tennessee, and Paducah, Kentucky, AIMSI provides professional engineering, management, technical and administrative support services and has formed the following team to provide both day-to-day project administrative support and public involvement and information:

Anna Corbett has more than 15 years experience in technical support and customer service, including extensive support to programs in the nuclear and chemical waste environment. Her career with AIMSI has included support to Rocky Flats programs including the Mixed Waste Focus Area, Technology Management program, and the Rocky Flats Site Technology Coordination Group.

Carla Sanda brings more than 15 years of experience in developing and executing stakeholder involvement activities. Prior to joining AIMSI as a subcontractor on this project, Ms. Sanda was a member of the Community Relations group at Rocky Flats. She served as team lead for waste management stakeholder involvement; project manager for the Rocky Flats Western Governors' Association Demonstration of Innovative Technologies; program manager for the Federal Facility Compliance Act stakeholder involvement; and was a member of the team tasked with drafting our national public involvement policy.

For the latest project information on the web go to:
www.rfcab.org/SALOP.html

Or contact AIMSI:
5460 Ward Road, Suite 370
Arvada, CO 80002
(303) 456-0884, Fax (303) 456-0858

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Project Passes Mid-Point:

- **Computer Model Chosen**
- **Key Inputs & Assumptions Identified**
- **Soil Sampling Protocols Reviewed**
- **Peer Review Team Mobilized**

September 1999

RADIONUCLIDE SOIL ACTION LEVEL OVERSIGHT PANEL

PROJECT UPDATE

PROJECT COMPRISED OF EIGHT TASKS

Work on review of the interim radionuclide soil action levels at Rocky Flats continues to move forward. As noted in the fact sheet entitled "*Planning for Tomorrow...Radionuclide Soil Action Levels at Rocky Flats*", the Radionuclide Soil Action Level Oversight Panel (RSALOP) was formed in response to concern voiced by members of the community at interim radionuclide soil action levels (RSALs) set for the Rocky Flats site. After receiving funding from the Department of Energy, the Panel contracted with *Risk Assessment Corporation* (*RAC*) to conduct an independent scientific assessment of the RSALs for Rocky Flats.

The project is organized into eight tasks. A final report will be published that will recap the study's findings and propose recommendations for a radionuclide soil action level (RSAL) at the Rocky Flats Site.

1. Cleanup Levels at Other Sites
2. Computer Models to Determine RSALS
3. Key Project Inputs & Assumptions
4. Methodology
5. Independent Calculation of RSALs for Rocky Flats
6. Sampling Protocol
7. Interaction with the Actinide Migration Panel
8. Public Involvement

Task 1: Cleanup Levels at Other Sites

Completed in April 1999, this report provided the Panel with a clear, unbiased evaluation and comparison of previously developed soil action levels for Rocky Flats and other sites around the world. The evaluation found that the soil action levels established for Rocky Flats are significantly higher than action or cleanup levels at other facilities, even when normalized to dose. However, the report provided a better understanding and clarification for the elevated levels and found that the calculation is strongly controlled by three basic parameters:

- Dose conversion factor (solubility class of plutonium)
- Mass loading (resuspension), and to a lesser degree
- Breathing rate

In nearly every case, differences in soil action levels between sites could be explained by the different assumptions made for one or more of the basic parameters. The Task 1 report identified the input model parameters that are of primary importance in determining soil action levels so that this information can be used as the study evolves.

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Task 2: Computer Models

The goal of this report, which was finalized in July 1999, was to discuss and compare environmental assessment programs that might be used for developing soil action levels for Rocky Flats. *RAC* scientists evaluated the following five computer programs for use in the project: **RESRAD, MEPAS, GENII, MMSOILS,** and **DandD**. The report discusses the pros and cons of each program and provides information on the four key elements that must be considered when developing soil action levels. Based upon extensive evaluation of the available computer codes, *RAC* representatives concluded that either the **RESRAD** or **GENII** program could be used. After further discussion with Panel members, it was decided to proceed with the use of the **RESRAD** program.

"Soil Action Levels Depend on Four Key Elements:

1. **How radioactive material is transported in the environment to people,**
2. **How people might be exposed to the radioactive materials (exposure scenarios,**
3. **How radiation dose to a person is assessed (radiation dosimetry), and**
4. **How radiation protection guidelines fit in (annual dose limits)."**

Task 3: Inputs & Assumptions

After months of discussions and input from Panel members, *RAC* representatives conducted a sensitivity analysis on the inputs and assumptions required for the use of RESRAD. Site-specific values were derived or uncertainty distributions were created for critical parameters emerging from the sensitivity analysis. The sensitivity of each parameters was then assessed using the built-in Monte Carlo-based sensitivity analysis packaged with the latest version of RESRAD. This report also includes a careful evaluation of scenarios for their applicability to potential future land uses. The report describes the process of scenario evaluation and discusses the scenarios chosen for the independent analysis of the Rocky Flats soil action levels. To develop meaningful and appropriate calculations for soil action levels at Rocky Flats, site-specific data were collected and will be used for all parameters that were revealed as sensitive to change and parameters that warranted adaptation. Primary efforts were 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. Values and distributions presented in this report will be used in the calibrated version of RESRAD, and values for soil action level and dose will be presented as distributions of possible values for each individual scenario.

Task 4: Methodology

"Methodology" is a topic that encompasses the project as a whole through ongoing dialogue with the Panel and Community regarding proposed methodologies employed in the study. Methodologies that may be considered and/or decided upon are discussed within reports specific to project tasks. Therefore, no separate report is being published on this task.

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Task 5: Independent Calculation of RSALs

Although this has been identified as "Task 5", this will be the final step in the study completed after running the RESRAD program using the Inputs & Assumptions decided upon from Task 3.

Task 6: Soil Sampling Protocols

RAC released the draft final report for Task 6 at the May 1999 Panel meeting. The document reviewed the current site sampling program and procedures as well as individual site sampling and analysis plans. It also went on to provide recommendations to the Panel for consideration in developing a sampling protocol for the site and discussed ten key elements that should be a part of any sampling protocol. Ultimately, *RAC* plans to recommend a specific soil sampling protocol to the Panel to determine the amount of contamination in soil following remediation. Task 6 is currently under revision, with an anticipated draft final report due in the fall of 1999.

TEN KEY ELEMENTS TO SAMPLING PROTOCOL

1. Data Quality Objectives
2. Multiple Radionuclide Consideration
3. Classification of Survey Units
4. Soil Sampling Depth
5. Sample Spacing & Methods
6. Small Areas of Elevated Activity
7. Surrogate Measurements
8. Number of Samples Based on Statistical Methods
9. Independent Confirmatory Investigations
10. Soil Sampling Quality Assurance

Task 7: Interaction with the Actinide Migration Panel

The Actinide Migration Panel is overseeing an effort begun by contractors at the Rocky Flats site in 1996. Comprised of a national task force, the group is drawing upon state-of-the-art knowledge throughout the scientific community on behavior and mobility of actinides in the environment. It is hoped that this group's efforts will help to provide information necessary to develop the best possible approach for the successful closure of the Rocky Flats site. *RAC* representatives, as well as numerous RSALOP members, attend regular Actinide Migration Panel meetings and are attempting to extrapolate any information gathered to assist in the independent review of the soil action levels for the Rocky Flats site. No separate formal report will be generated on this Task.

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Task 8: Public Involvement

A public involvement strategy has been developed to provide regular updates to the community on the progress of this study. Panel members meet the second Thursday of each month with *RAC* representatives to review project findings and work with the contractor to set and determine criteria for key components of the study. You are invited to attend the monthly meetings, which are held from 4-7 p.m. at the Broomfield City Building, One Descombes Drive, Broomfield, CO. Three public meetings are scheduled to receive input from the community throughout all phases of the technical review. In addition, with some up-front planning, Panel members can provide briefings to community groups or interested parties. Please contact Carla Sanda at 303-277-0753 for information regarding public meetings, project updates, or speakers for your group.

PEER REVIEW TEAM MOBILIZED

To enhance the quality and credibility of this effort, the Panel has formed a Peer Review Team comprised of five nationally recognized experts with backgrounds related to this effort. Team members are tasked with reviewing and commenting on each draft final report produced for this project. Comments are then forwarded to *Risk Assessment Corporation*, who reviews and provides feedback on the peer review input. Members of the Panel then look at the Peer Review input coupled with the feedback from *RAC* to assure that Panel members concur with comment resolution.

We're on the Web:
www.rfcab.org/SALOP.html

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RAC **concludes that using its methodology and assumptions "indicates a technically based RSAL for 239 & 240 Pu in soil at Rocky Flats of 35 pCi/gram. For uranium, a technically derived RSAL...would be 10 pCi/gram."**

(See back page for full statement.)

February 2000

RADIONUCLIDE SOIL ACTION LEVEL OVERSIGHT PANEL

PROJECT UPDATE

PROJECT COMPRISED OF EIGHT TASKS

Work on review of the interim radionuclide soil action levels at Rocky Flats is nearing its scheduled completion date of March 31, 2000. As noted in the fact sheet entitled "*Planning for Tomorrow...Radionuclide Soil Action Levels at Rocky Flats*", the Radionuclide Soil Action Level Oversight Panel (RSALOP) was formed in response to concern voiced by members of the community at interim radionuclide soil action levels (RSALs) set for the Rocky Flats site. After receiving funding from the Department of Energy, the Panel contracted with *Risk Assessment Corporation* (*RAC*) to conduct an independent scientific assessment of the RSALs for Rocky Flats.

The project is organized into eight tasks. In addition, a final summary report will be published to recap the study's findings and propose recommendations for a radionuclide soil action level (RSAL) at the Rocky Flats Site.

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- Dose conversion factor (solubility class of plutonium)
- Mass loading (resuspension), and to a lesser degree
- Breathing rate

In nearly every case, differences in soil action levels between sites could be explained by the different assumptions made for one or more of the basic parameters. The Task 1 report identified the input model parameters that are of primary importance in determining soil action levels so that this information can be used as the study evolves.

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Task 3: Inputs & Assumptions

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Primary efforts were 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. Values and distributions presented in this report will be used in the calibrated version of RESRAD, and values for soil action level and dose will be presented as distributions of possible values for each individual scenario.

Task 4: Methodology

"Methodology" is a topic that encompasses the project as a whole through ongoing dialogue with the Panel and Community regarding proposed methodologies employed in the study. Methodologies that may be considered and/or decided upon are discussed within reports specific to project tasks. Therefore, no separate report is being published on this task.

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Task 5: Independent Calculation of RSALs

Although this report has been identified as "Task 5", it is actually the final step in the study to be completed after running the RESRAD program using the Inputs & Assumptions decided upon from Task 3. The Draft Final Task 5: Independent Calculation presents the results of *RAC's* independent assessment and describes the calculations and results of the soil action levels for the seven exposure scenarios identified in the Task 3 report.

Task 6: Soil Sampling Protocols

RAC released the draft final report for Task 6 at the December 1999 Panel meeting. The document reviewed the current site sampling program and procedures as well as individual site sampling and analysis plans. It also went on to provide recommendations to the Panel for consideration in developing a sampling protocol for the site and discussed ten key elements that should be a part of any sampling protocol. The report concluded that the MARSSIM guidance provides the most comprehensive approach currently available for the development of radiological surveys and recommended that the final status surveys conducted at Rocky Flats follow the general principles contained therein.

TEN KEY ELEMENTS TO SAMPLING PROTOCOL

1. Data Quality Objectives
2. Multiple Radionuclide Consideration
3. Classification of Survey Units
4. Soil Sampling Depth
5. Sample Spacing & Methods
6. Small Areas of Elevated Activity
7. Surrogate Measurements
8. Number of Samples Based on Statistical Methods
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Task 7: Interaction with the Actinide Migration Panel

The Actinide Migration Panel is overseeing an effort begun by contractors at the Rocky Flats site in 1996. Comprised of a national task force, the group is drawing upon state-of-the-art knowledge throughout the scientific community on behavior and mobility of actinides in the environment. It is hoped that this group's efforts will help to provide information necessary to develop the best possible approach for the successful closure of the Rocky Flats site. *RAC* representatives, as well as numerous RSALOP members, attend regular Actinide Migration Panel meetings and are attempting to extrapolate any information gathered to assist in the independent review of the soil action levels for the Rocky Flats site. No separate formal report will be generated for this Task.

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Task 8: Public Involvement

A public involvement strategy was developed and implemented to provide regular updates to the community on the progress of this study. Panel members meet the second Thursday of each month with *RAC* representatives to review project findings and work with the contractor to set and determine criteria for key components of the study. You are invited to attend the monthly meetings, which are held from 4-7 p.m. at the Broomfield City Building, One Descombes Drive, Broomfield, CO. Three public meetings were scheduled at key points to receive input from the community throughout all phases of the technical review. In addition, with some up-front planning, Panel members can provide briefings to community groups or interested parties. Please contact Mary Harlow, Panel Co-Chair, at 303-430-2400, Ext 2174, for additional details.

PEER REVIEW TEAM COMPLETES ITS WORK

To enhance the quality and credibility of this effort, the Panel formed a Peer Review Team comprised of five nationally recognized experts with backgrounds related to this effort. Team members were tasked with reviewing and commenting on each draft final report produced for this project. Comments were then forwarded to *Risk Assessment Corporation*, who reviewed and provided feedback on the peer review input at the Panel meetings. Members of the Panel then looked at the Peer Review input coupled with the feedback from *RAC* to assure that Panel members concurred with *RAC's* comment resolution.

RAC'S CONCLUSIONS REGARDING RSALS FOR ROCKY FLATS

"*RAC's* task was to evaluate the RSALs adopted for Rocky Flats in 1996, to develop a methodology for independently determining RSALs, and to calculate RSALs for Rocky Flats by applying this methodology. We conclude that applying our method to the exposure scenarios approved by the Oversight Panel, using 15 millirem as a dose limit, and assuming a probability level of 10%, indicates a technically based RSAL for 239 and 240 Pu in soil at Rocky Flats of 35 pCi g^{-1}. For uranium, a technically derived RSAL using our methodology and assumptions would be 10 pCi g^{-1}".

(Quoted from the "Conclusions" Section of *RAC's* Final Project Summary)

FINAL PUBLIC MEETING

Complete project recommendations will be presented to the community-at-large at the final public meeting scheduled for March 23, 2000, from 7:00-9:00 p.m. at the Broomfield City Center - Council Chambers, One Descombes Dr., Broomfield, CO. Please join the Panel and representatives from the Department of Energy, the Colorado Department of Public Health and Environment, and the Environmental Protection Agency to learn more about this important issue and help pave the way to a future that will be protective of the communities surrounding the Rocky Flats site.

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