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"
RAC
Risk Assessment Corporation

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in Partial Fulfillment of Contract between RAC and the Rocky Flats Citizen's Advisory Board

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 5% to 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% to 95% 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.

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.

CONTENTS

GENERAL SUMMARY iii
CONTENTS vii
FIGURES ix
TABLES xii

1. INTRODUCTION 1-1

2. BACKGROUND OF THE STUDY 2-1

3. ISOTOPIC RATIOS FOR PLUTONIUM, AMERICIUM, AND NEPTUNIUM 3-1

4. RECENT SPATIAL DISTRIBUTIONS OF PLUTONIUM IN THE SOIL NEAR THE ROCKY FLATS SITE 4-1
   4.1. A Spatial Model of the Plutonium-239 Concentration in Soil 4-1

5. ESTIMATING CONCENTRATIONS OF PLUTONIUM IN AIR 5-1
   5.1. Model of Resuspension and Atmospheric Transport 5-2
   5.2. Nonlinear Regression based on Air Monitoring Data to Estimate Resuspension Parameters 5-3
   5.3. Resuspension from Unvegetated Soil After a Fire 5-15
   5.4. Resuspension Parameters for the Scenarios 5-17
   5.5. Probabilistic Incorporation of Future Fires into the Scenarios 5-17

6. URANIUM METHODOLOGY 6-1
   6.1. Uranium Mass Loading 6-1
   6.2. Implications of a Fire 6-2
   6.3. Sum-of-Ratios for Uranium 6-5

7. ALTERNATE CALCULATION 7-1
   7.1. Truncation of Soil Concentration at a Specified Contour 7-1
   7.2. Modification of the Fire Model 7-1
   7.3. Calculations with the Alternative Method 7-2
   7.4. Implications for the Definition of a Soil Action Level 7-4

8. THE ROLE OF RESRAD 8-1

9. METHOD OF RSAL PRESENTATION 9-1

10. CONSIDERATIONS IN SELECTING A RADIONUCLIDE SOIL ACTION LEVEL . 10-1
    10.1. Criteria That Have been Taken into Account in *RAC*'s Analysis of RSALs ... 10-1
    10.2. Criteria That Have Not Been Taken into Account in *RAC*'s Analysis of RSALs 10-3

11. RESULTS 11-1
11.1.Introduction to Plutonium RSALs 11-1
11.2.Important Pathways for Plutonium Isotope Simulations 11-2
11.3. Introduction to Uranium RSALs 11-3
11.4. Scenario Probability Curves for Plutonium 11-4
11.5. Scenario Probability Curves for Uranium 11-18

12. DISCUSSION AND RECOMMENDATIONS FOR FURTHER RESEARCH 12-1
12.1.Heterogeneity of Plutonium Concentrations 12-1
12.2.Uncertainty 12-2
12.3. Scenarios 12-4
12.4. Future Grass Fire 12-5
12.5. Other Recommendations 12-5

13. CONCLUSIONS 13-1

14. REFERENCES 14-1

APPENDIX A: COMPUTATIONAL DETAILS

APPENDIX B: ALTERNATIVE GROUNDWATER DOSE CALCULATION FROM MEASUREMENTS BY LITAOR

APPENDIX C: EVALUATION OF BACKGROUND CONCENTRATIONS OF PLUTONIUM IN THE ENVIRONMENT AROUND ROCKY FLATS

APPENDIX D: THE EFFECT OF TIME ON THE RSAL

APPENDIX E: RISKS ASSOCIATED WITH A 15 MREM ANNUAL DOSE LIMIT

FIGURES

3-1. Relative kinetics of plutonium, americium, and neptunium in Rocky Flats soil from 1971 to 2050 3-2

4-1. Regression curve based on data summarized by Webb (1996) 4-4

4-2. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model 4-7

4-3. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 Area 4-9

4-4. Contours of approximate ^{239}Pu concentration in soil (becquerels per kilogram) based on the spatial distribution model described in the text 4-10

5-1. Model-predicted plutonium concentrations (Bq m^{-3}) plotted against observations of $^{239+240}$Pu at the Rocky Flats S-series samplers 5-8

5-2. Cumulative probability plot of predicted-to-observed ratios from the regression on the sampler data 5-9

5-3. Distribution of points from the bootstrap regression of predicted air concentration on the sampler data 5-10

5-4. Empirical cumulative distribution function of estimated soil flux with fitted lognormal distribution 5-12

5-5. Sampler-efficiency curve fitted to data discussed by Rope et al. (1999) 5-14

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 5-16

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 6-3

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 6-4

8-1. Schematic illustration of a scenario simulation using a PERL script with the RESRAD computational module 8-2

9-1. Generic distribution showing the relationship between probability of exceeding the dose limit as a function of plutonium concentration 9-1

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 11-3

11-2. Curve representing the probability of exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario 11-4

11-3. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE Open space user scenario 11-6

11-4. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE office worker scenario 11-8

11-5. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario 11-10

11-6. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario 11-12

11-7. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* infant of rancher scenario 11-14

11-8. Probability of the total dose exceeding the 15 and 85 mrem dose limits for the *RAC* site industrial worker scenario 11-16

11-9. Probability of the total dose exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario 11-19

11-10. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario 11-21

11-11. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario 11-23

13-1. Composite graphic illustrating the most restrictive scenarios and showing a region centered at a soil action level of 35 pCi g^{-1} 13-2

TABLES

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

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

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

5-1. Summary of Parameters and Numeric Results for Resuspension Fluxes 5-2

5-2. Sampling Locations with Predicted and Observed Plutonium Air Concentrations 5-7

5-3. Regression Parameter Estimates for Resuspension Flux 5-8

5-4. Plutonium-239+240 Air and Soil Concentrations for Rocky Flats Scenarios 5-17

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..... 5-19

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..... 7-3

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}) 11-5

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}) 11-7

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}) 11-9

11-4. Percentiles from the Distribution of Individual Plutonium Isotope RSALs for the *RAC* Rancher Scenario, 15 mrem dose limit (pCi g^{-1})..... 11-11

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})..... 11-13

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})..... 11-15

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}) 11-17

11-8. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the DOE/EPA/CDPHE Resident Scenario (pCi g^{-1}) 11-20

11-9. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the *RAC* Resident Rancher Scenario (pCi g^{-1}) 11-22

11-10. Percentiles from the Distribution of Individual Uranium Isotope RSALs for the *RAC* Child of Rancher Scenario (pCi g^{-1}) 11-24

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 → ^{241}Am → ^{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 → ^{241}Am → ^{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.

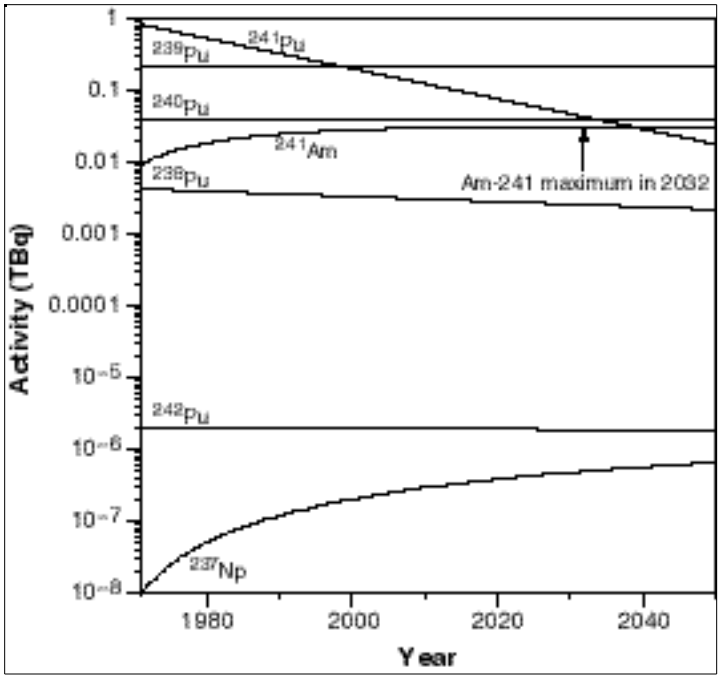

Figure 3-1. Relative kinetics of plutonium, americium, and neptunium in Rocky Flats soil from 1971 to 2050. The activity proportions correspond to 100 g of plutonium with isotope mass ratios given by Krey et al. (1976).

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[1], 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

[1] 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.

the sampling depth and an approximate time when the samples were taken (one set of 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).

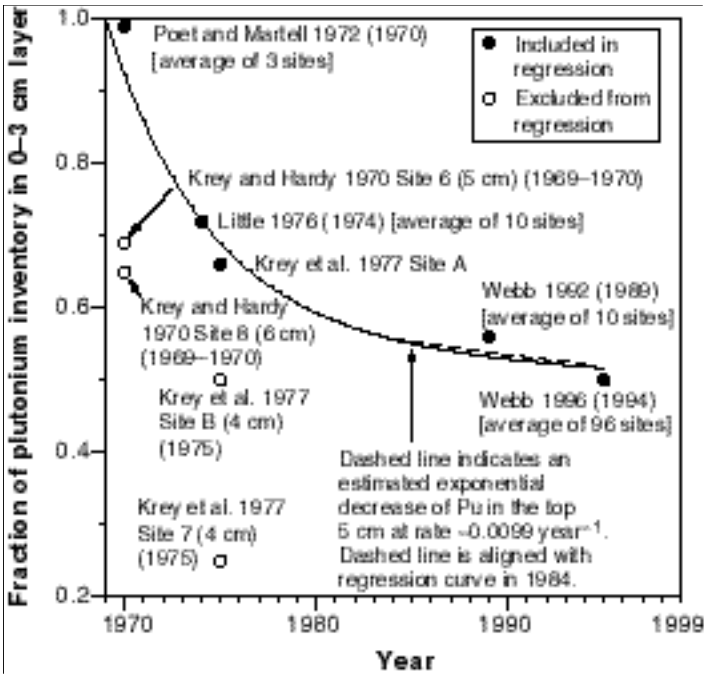

Figure 4-1. Regression curve based on data summarized by Webb (1996). The curve represents the equation $y = a\exp(-b_1 t) + (1-a)\exp(-b_2 t)$, with nonlinear least squares–determined parameters a = 0.54769, b_1 = −0.00255, and b_2 = −0.1825. The black circles represent the data used for the regression, 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.[2]

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

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

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:

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

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

the 0–z_d-cm layer, we may ress it as

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

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:

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

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

$$\rho(z) = 0.79 \cdot 10^{-3} z^{0.24} \quad \mathrm{kg\,cm^{-3}} \,. \quad (4\text{-}4)$$

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.

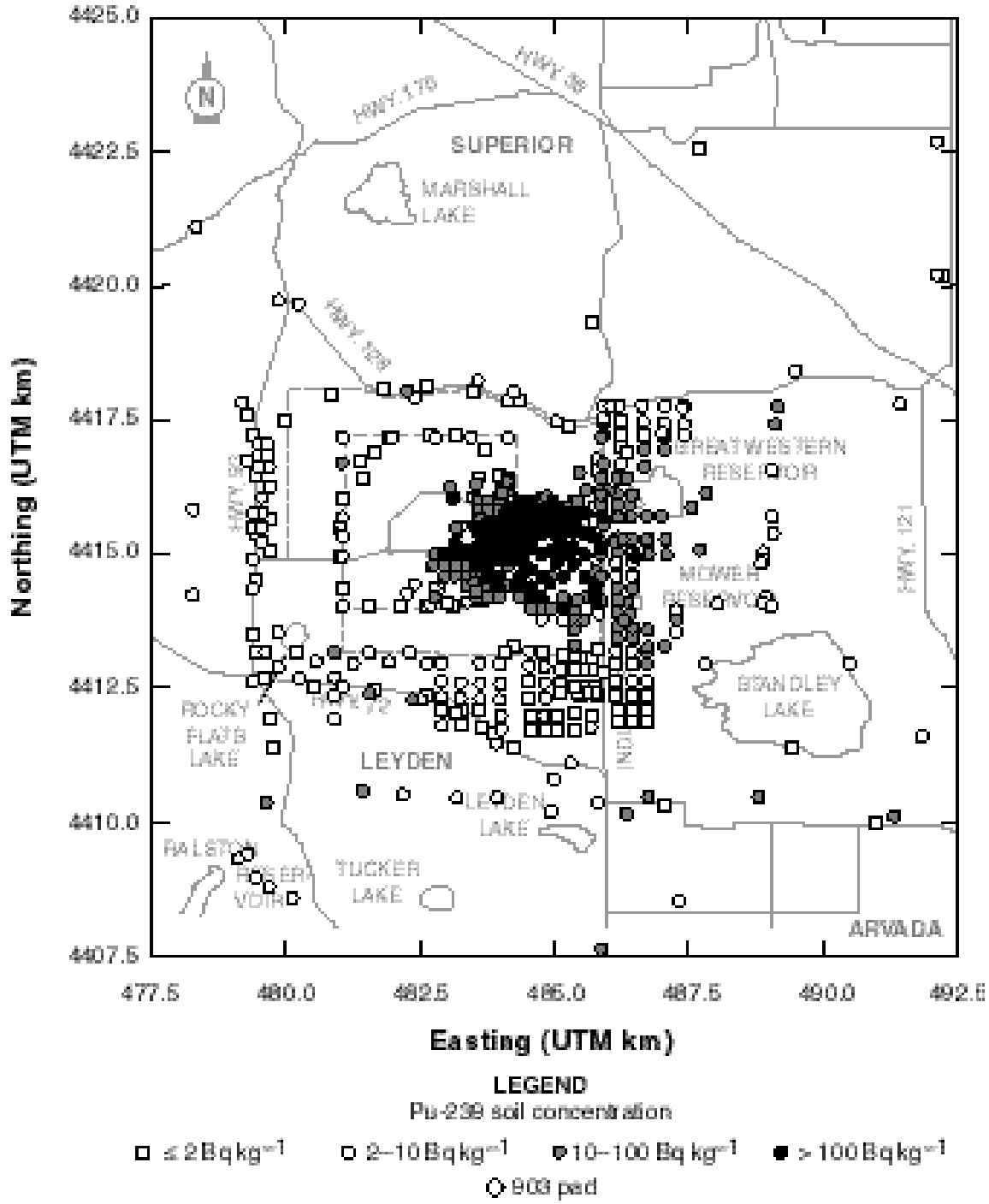

Figure 4-2. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model. The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), Ripple et al. (1994), and one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

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[3] 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 60º, 90º, and 120º 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.

[3] 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.

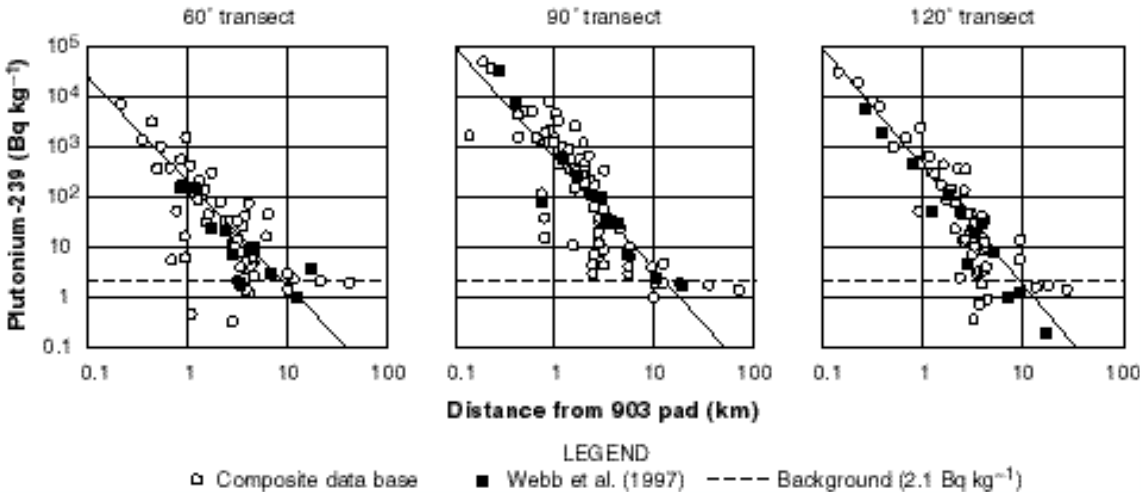

Figure 4-3. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 pad. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols. 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.

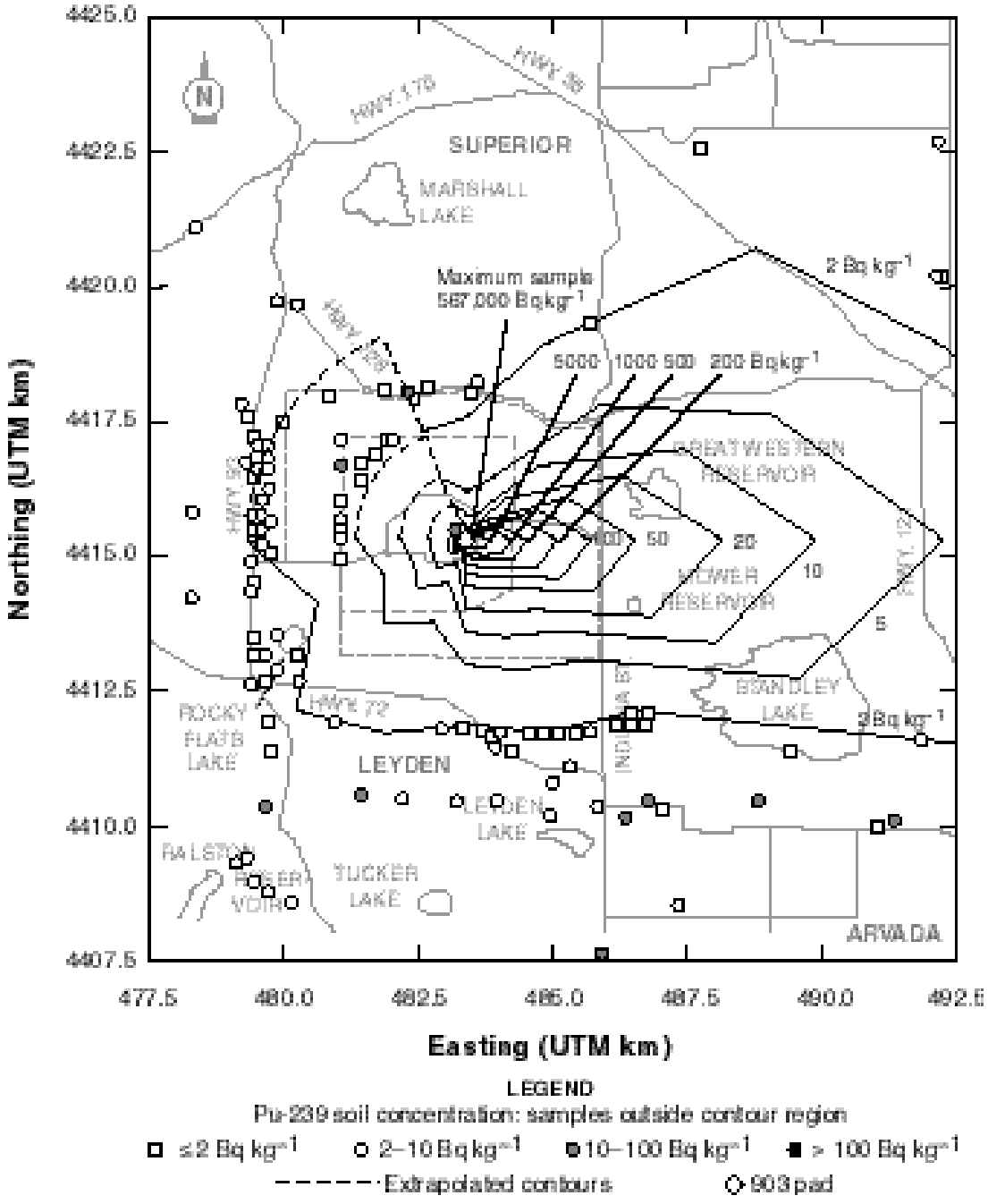

Figure 4-4. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text. Dashed parts of the contours indicate extrapolation where coverage was insufficient for fitting power functions. In these regions and outside the 2 Bq kg^{-1} 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.

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})	$3.3 \times 10^{-5} –$ 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

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

(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

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

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 ( ) 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

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

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 $\bar{\chi}(m \mid F_0, \gamma)$ 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 $\bar{\chi}_{mt}$. (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

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

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 $r_{mt} = \ln \bar{\chi}_{mt} - \bar{\chi}(m \mid \hat{F}_0, \hat{\gamma})$ 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:

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

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 $\mathrm{GSD}_{\mathrm{resid}} = 3.03$, 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

			Plutonium-239+240 in air (Bq m^{-3})			
				Observed		
Station	Distance (km)	Angle (°)	Predicted	1992	1993	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}

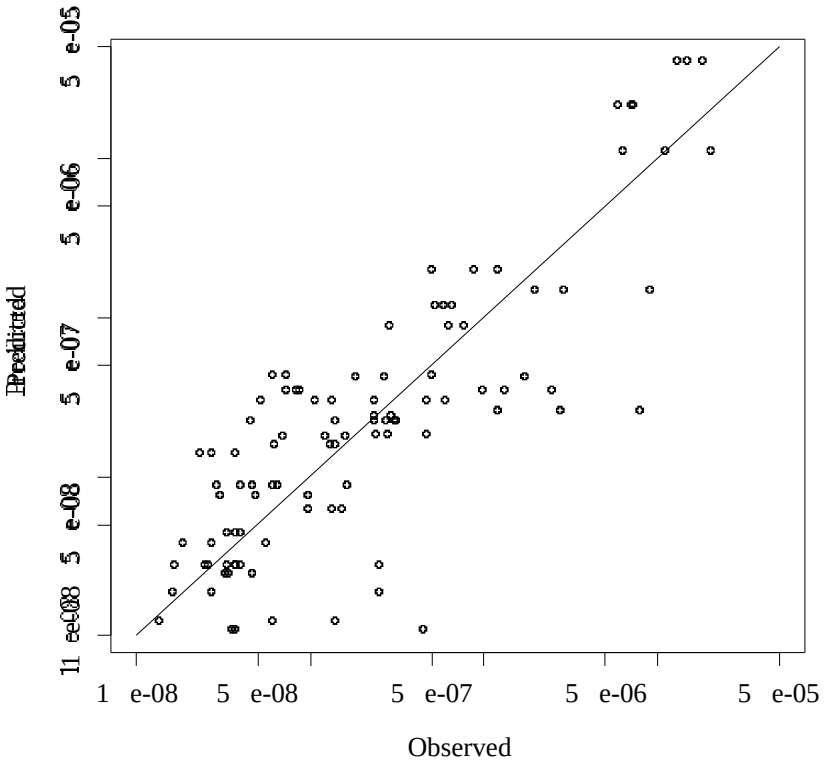

Figure 5-1. Model-predicted plutonium concentrations (Bq m^{-3}) plotted against observations of $^{239+240}Pu$ at the Rocky Flats S-series samplers. The units for both axes are Bq m^{-3}. The line is the locus of points for which the prediction equals the observed value.

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		

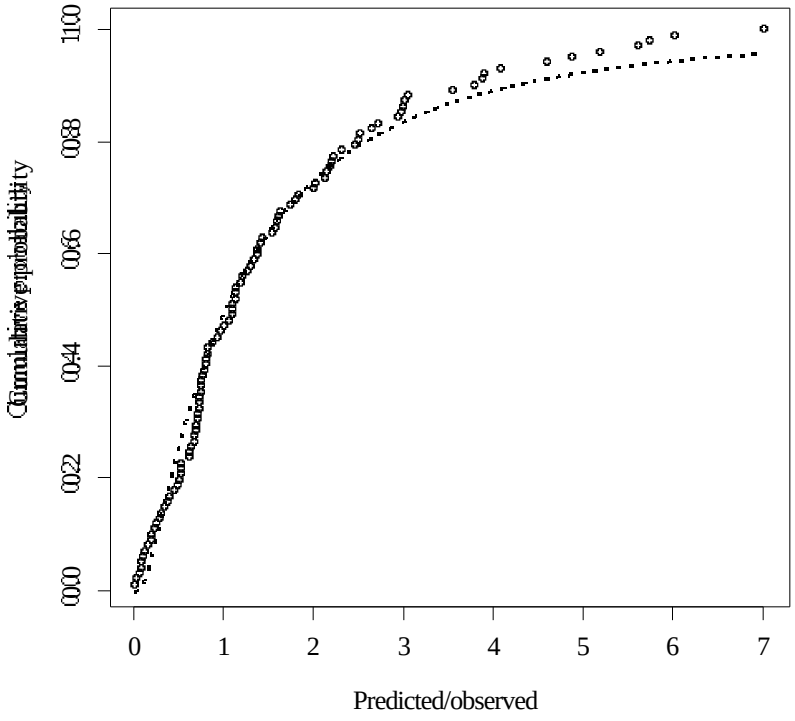

Figure 5-2. Cumulative probability plot of predicted-to-observed ratios from the regression on the sampler data. The points represent the empirical distribution function of the P/O ratios based on the logarithmic residuals. The dashed curve is the lognormal distribution with geometric mean 1.00 and geometric 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 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:

$$\chi(x, y, z) \quad F \cdot G(x, y, z; C_{\text{soil}}) . \tag{5-6}$$

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.

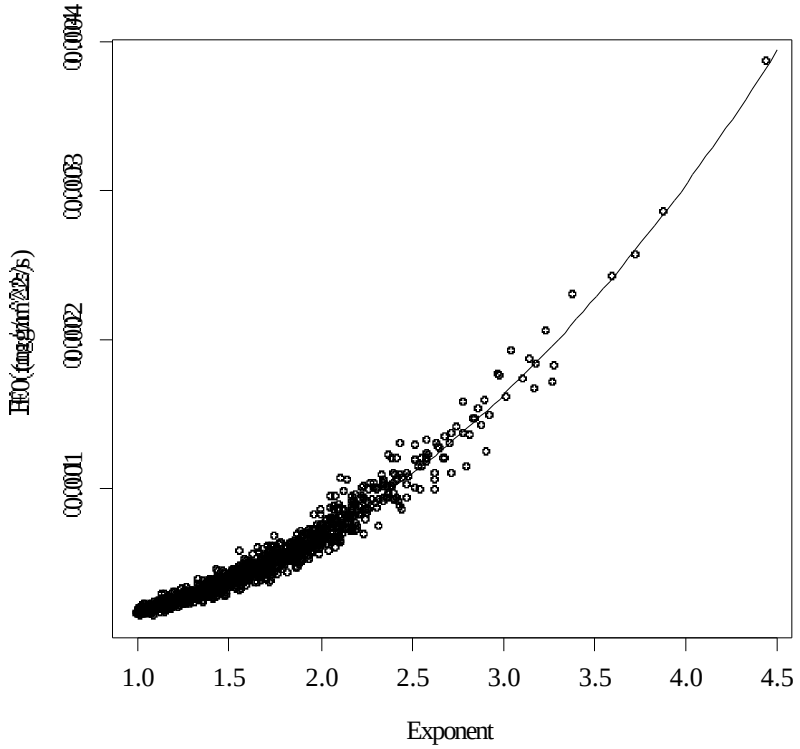

Figure 5-3. Distribution of points (Exponent = $1+\gamma$, F_0) from the bootstrap regression of predicted air concentration on the sampler data. The units of the reference flux F_0 are mg m^{-2} s^{-1}, and the exponent $1+\gamma$ is dimensionless. The fitted curve is the cubic polynomial $f(\theta) = c_0 + c_1\theta + c_2\theta^2 + c_3\theta^3$, with $\theta = 1+\gamma$ and coefficients $c_0 = 5.688 \times 10^{-5}$, $c_1 = -6.749 \times 10^{-5}$, $c_2 = 1.795 \times 10^{-4}$, $c_3 = 6.032 \times 10^{-6}$.

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

$$\text{GSD} = \exp\sqrt{(\ln 3.03)^2 + (\ln 1.16)^2} = 3.06 \qquad (5\text{-}7)$$

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.

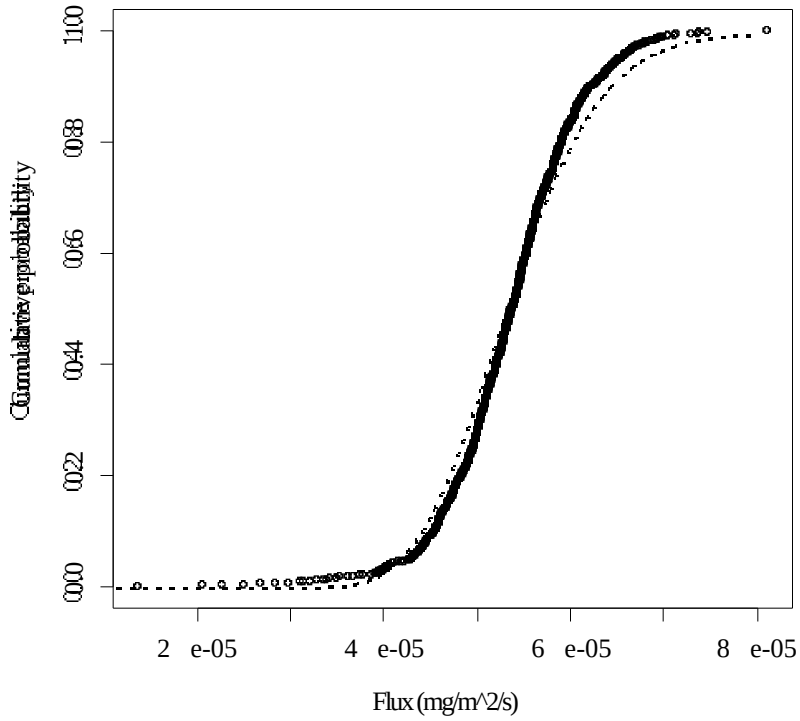

Figure 5-4. Empirical cumulative distribution function of estimated soil flux (solid curve) with fitted lognormal distribution (dashed curve). The parameters of the lognormal distribution are geometric mean: 5.31×10^{-5} mg m^{-2} s^{-1}, and geometric standard deviation: 1.16. The bootstrap median (5.38×10^{-5} 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

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

$$E = \frac{\int_0^\infty D^2 \varepsilon(\sqrt{\rho} D) f_c(D)\, dD}{\int_0^\infty D^2 f_c(D)\, dD} \qquad (5\text{-}9)$$

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 ρ = 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 $GM_c = GM_m \exp(-2 \ln^2 GSD)$ (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 $15/\sqrt{2.6} = 9.3$ μm 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.

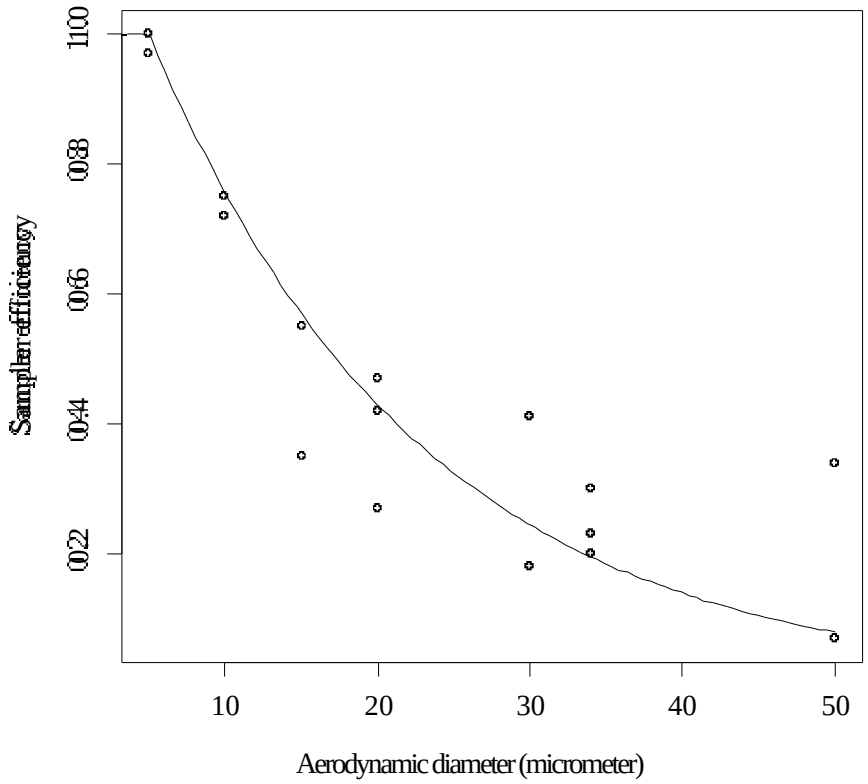

Figure 5-5. Sampler-efficiency curve fitted to data discussed by Rope et al. (1999). Some of the scatter in the plotted points is accounted for by different orientations of the device to the airflow. The function has the form $\varepsilon(D) = \exp(-b(D-5))$ (b = 0.0563 μm^{-1}) for aerodynamic diameter $D \geq 5$ μm 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

$$R = -0.4\, p u_* S_f \qquad (5\text{-}14)$$

(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 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		
				Unvegetated soil	Soil	
(Bq kg^{-1})	Site workers	0.59	330	5.6×10^{-7}	9.9×10^{-5}	17
	DOE recreational user	0.30	90	2.8×10^{-5}	5.4×10^{-3}	8.9×10^{3}

Rancher, DOE resident,

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

$$54 / 75 = 0.72 , \qquad (5\text{-}15)$$

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

$$p = \frac{6.1 \times 10^9 \text{ m}^2}{\qquad} \cdot \qquad (5\text{-}16)$$

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]

YearArea (km ^{2})CauseYearArea (km^2)Cause19244.65Unknown19710.85Human19321.82Human19761.04Human19322.02Railroad19761.01Human19341.58Railroad19781.62Human19382.02Railroad19784.86Human19394.08Human19791.26Human19430.93Human19801.93Human19444.82Lightning198011.06Human19440.69Lightning19800.73Human19520.72Human19804.05Lightning19524.91Lightning19850.95Lightning19531.15Human19851.21Lightning19541.04Human198811.33Human19562.89Lightning19883.06Human19580.68Lightning198810.00Human19602.45Human19880.81Lightning19602.67Lightning19897.30Human19600.61Lightning19891.47Lightning19624.08Human19897.96Lightning19623.03Human19890.40Human19620.57Human19900.57Lightning19630.74Human19931.11Lightning19630.89Lightning19944.94Lightning19661.90Human19941.50Lightning19682.99Lightning19961.50Lightning197110.66

Human19980.53Human19710.50Lightning19981.93Human [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:

$$\Phi = r_F / r_G . \qquad (5\text{-}17)$$

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 $\phi \quad \Phi_0 f_B g \exp(-\alpha(m-1))$. If $\phi > \Phi$, set $\Phi \quad \phi$ and $m^* \leftarrow m$.

(4) If $m < 1000$, $m \quad m+1$ and return to step (2).

(5) Otherwise, make the adjustment $\Phi \quad \Phi \exp(\alpha(m^*-1))$ 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 Φ = 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

$$C_a = MLC_s \qquad (6\text{-}1)$$

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,[4] 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

$$P = 6.1 \times 10^{9}\ m^{-2} \times$$

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 area. In addition, the modifying factors described in Section 5.5.2 were not considered here. The

[4] Personal communication with N.D. Chick, Colorado Department of Health, Air Pollution Division, June 19, 1992.

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.

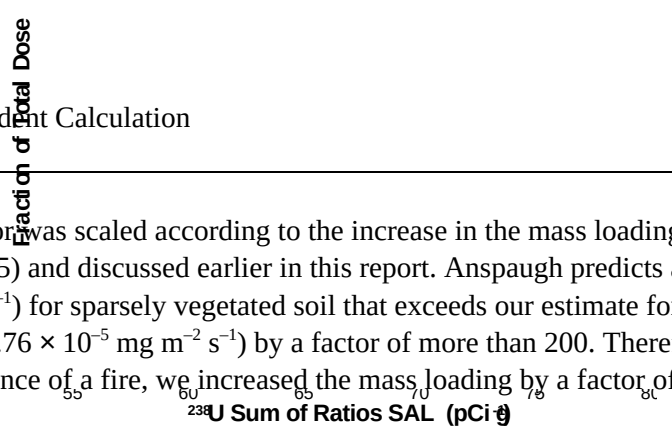

Figure 6-12. Fraction of the total dose at the time of maximum dose for different pathways of exposure as a function of the ^{238}U RSAL with the groundwater pathway turned off. The sum-of-ratios calculation is incorporated into the ^{238}U RSAL. Note that soil ingestion and inhalation make up only a 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 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 6-13. 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.

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

$$C_a^* = g\,(C_a + \chi\,?\Delta r)\exp(-\alpha\,t) = C_a\,g\,(1 + \chi\,?\Delta r / C_a)\exp(-\alpha\,t) = C_a\Phi \qquad (7\text{-}1)$$

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 $\phi \quad (1 + \chi\,?\Delta r / C_a)\exp(-\alpha\,(m-1))\,?g$. If $\phi > \Phi$, set $\Phi \quad \phi$ and $m^* \leftarrow m$.

(4) If $m < 1000$, $m \quad m + 1$ and return to step (2).

(5) Else set $t \quad m - 1$ 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}80002160.31.28 × 10^{-5}3.46 × 10^{-4}85%60001620.31.05 × 10^{-5}2.83 × 10^{-4}70.5%40001080.47.72 × 10^{-6}2.08 × 10^{-4}51.5%2000540.54.58 × 10^{-6}1.24 × 10^{-4}20.1%1000270.62.62 × 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}):

$$C_a = C_s \quad \text{ML} \quad \text{AF} \quad (8\text{-}1)$$

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 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.

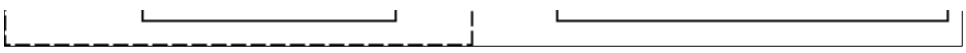

Figure 8-1. Schematic illustration of a scenario simulation using a PERL script with the RESRAD computational module. Prior calculations with other programs give a product ML AF (mass loading factor times area factor) appropriate to the scenario and the spatial distribution of plutonium in the soil. 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

EMBED Equation.2 (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 (_m) 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 concentration 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 AF_R is also calculated externally by the script program. By inputting the value ML AF / 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 AF_R and substitution into the input file. Thus, the results are independent of the RESRAD area factor 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:

$$S = \left(\frac{r_1}{\mathrm{RSAL}_1} + \cdots + \frac{r_N}{\mathrm{RSAL}_N} \right) C_{\text{Pu-239+240}} \qquad (8\text{-}3)$$

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 RSAL_1, , 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.

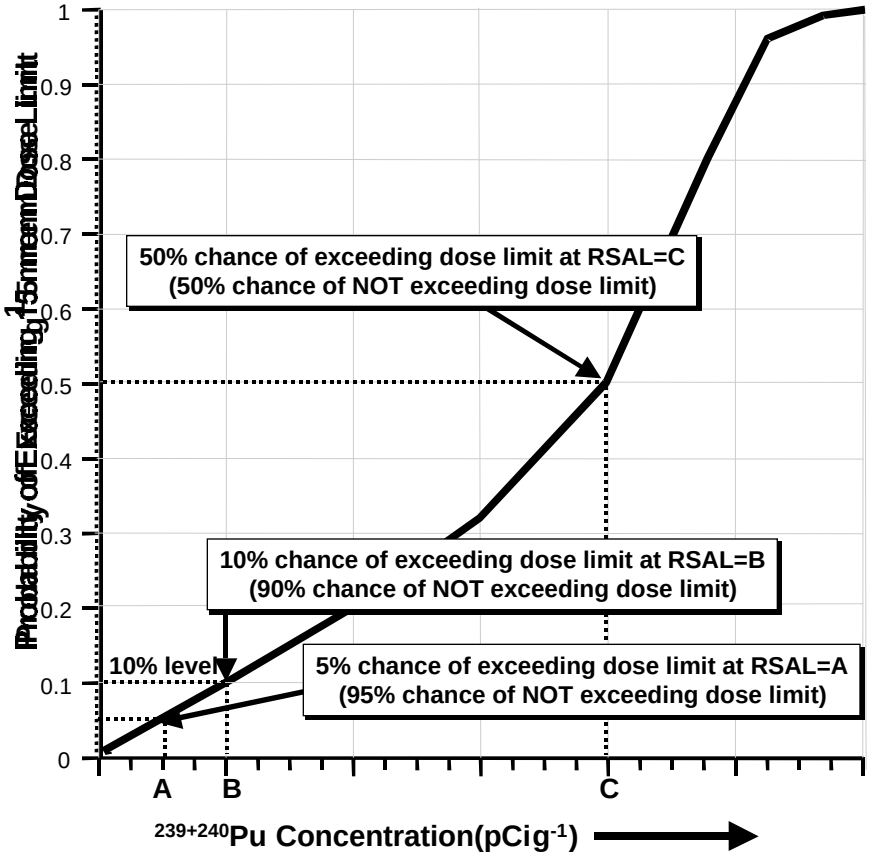

Figure 9-1. Generic distribution showing the relationship between probability of exceeding the dose limit as a function of plutonium concentration. As the contaminant concentration in soil increases, so does the probability of exceeding the dose limit.

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 $5 \cdot 10^{-4}$, with a large uncertainty range from about $2 \cdot 10^{-5}$ to $15 \cdot 10^{-3}$. The risk associated with exposure for 30 years at 15 mrem y^{-1} was approximately $2.3 \cdot 10^{-4}$ with a range from $1 \cdot 10^{-5}$ to $7 \cdot 10^{-3}$. 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 $7.5 \cdot 10^{-6}$ to $2.3 \cdot 10^{-4}$. 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.

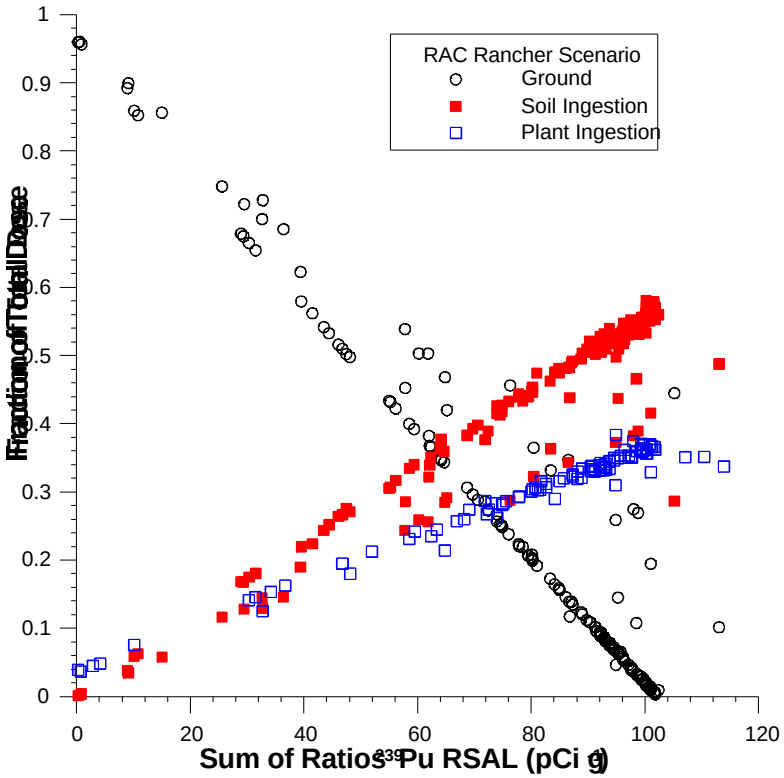

Figure 11-14. Fraction of the total dose for the rancher scenario as a function of the $^{239+240}$Pu sum-of-ratios RSAL for the three primary exposure pathways (inhalation, soil ingestion, and plant ingestion).

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.

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:

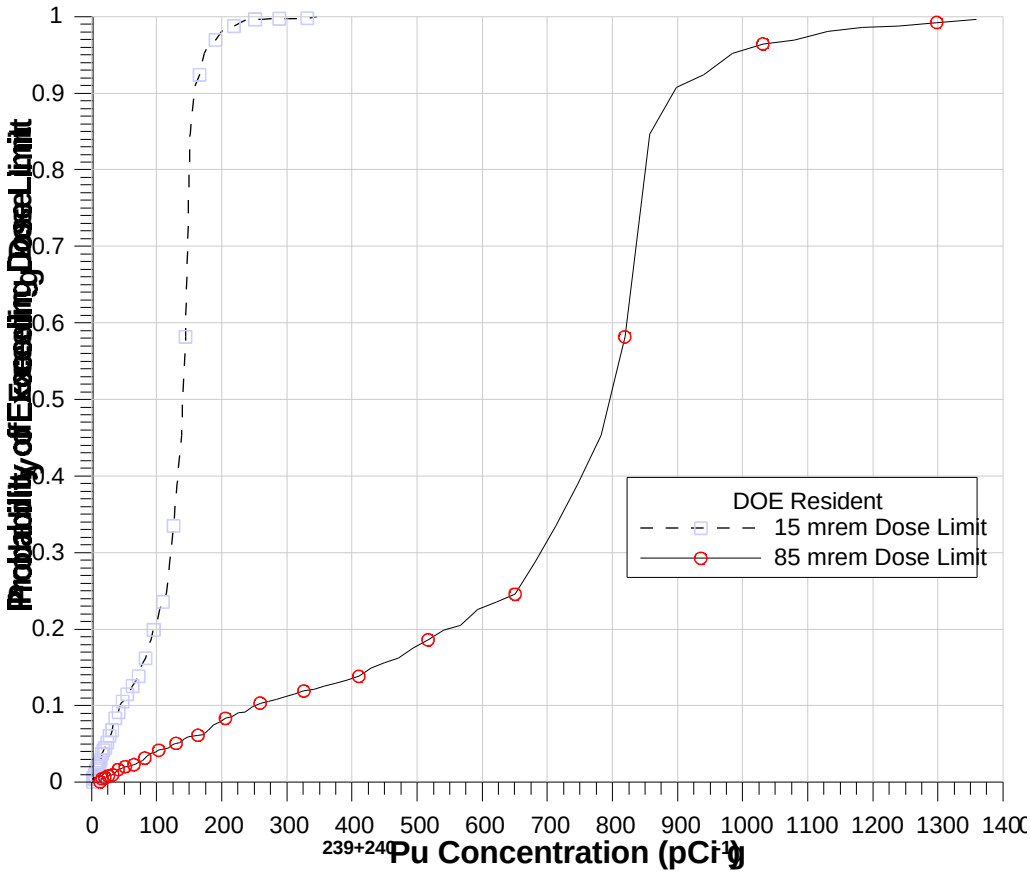

Figure 11-15. Curve representing the probability of exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

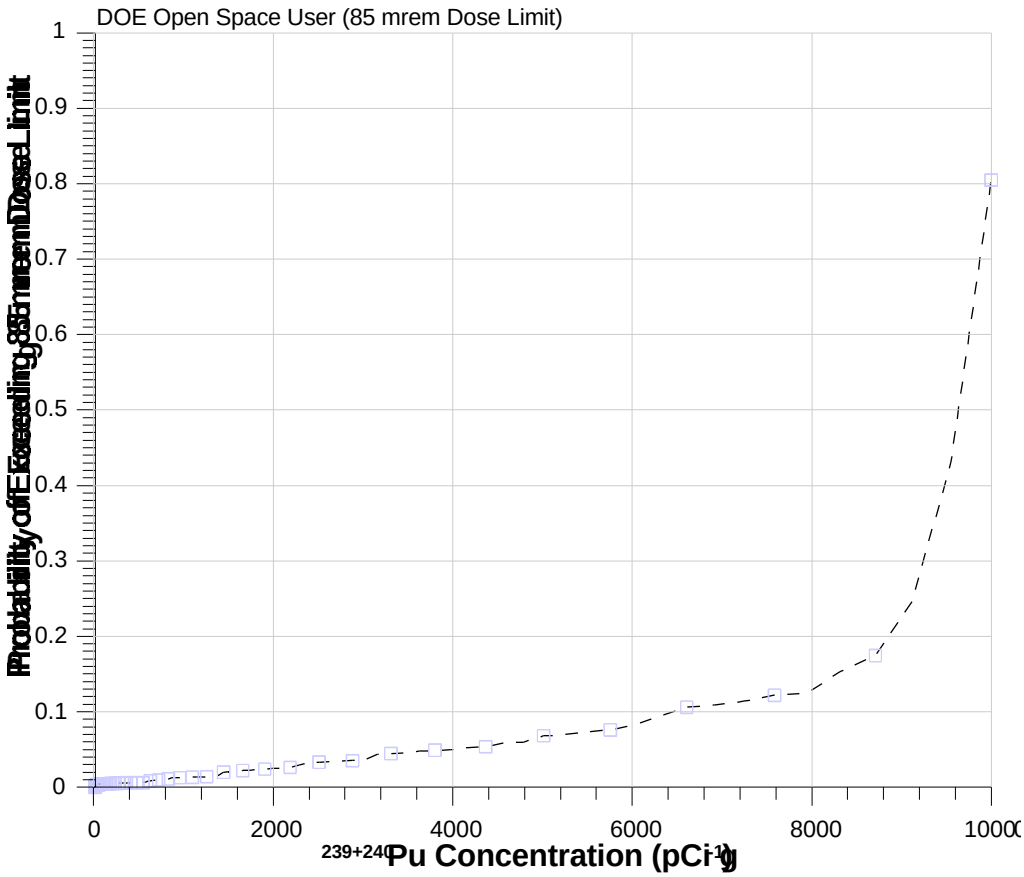

Figure 11-16. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE Open space user scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

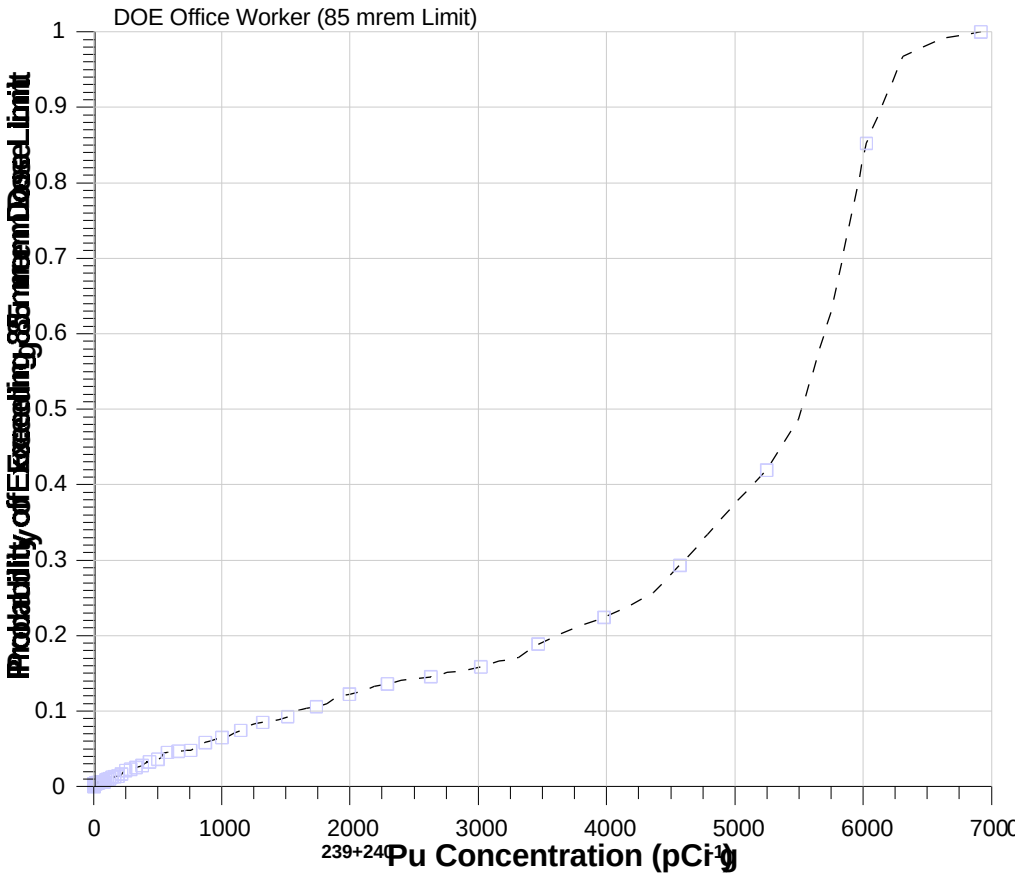

Figure 11-17. Probability of the total dose exceeding the 85 mrem dose limit for the DOE/EPA/CDPHE office worker scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

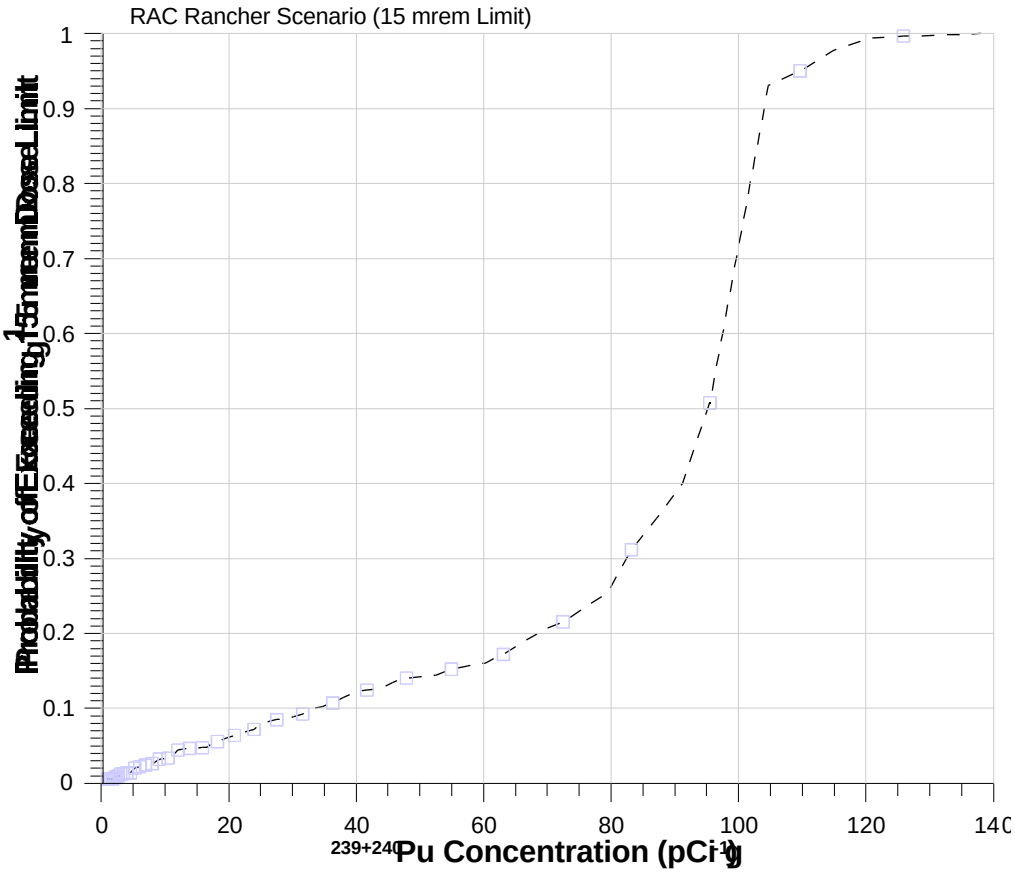

Figure 11-18. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

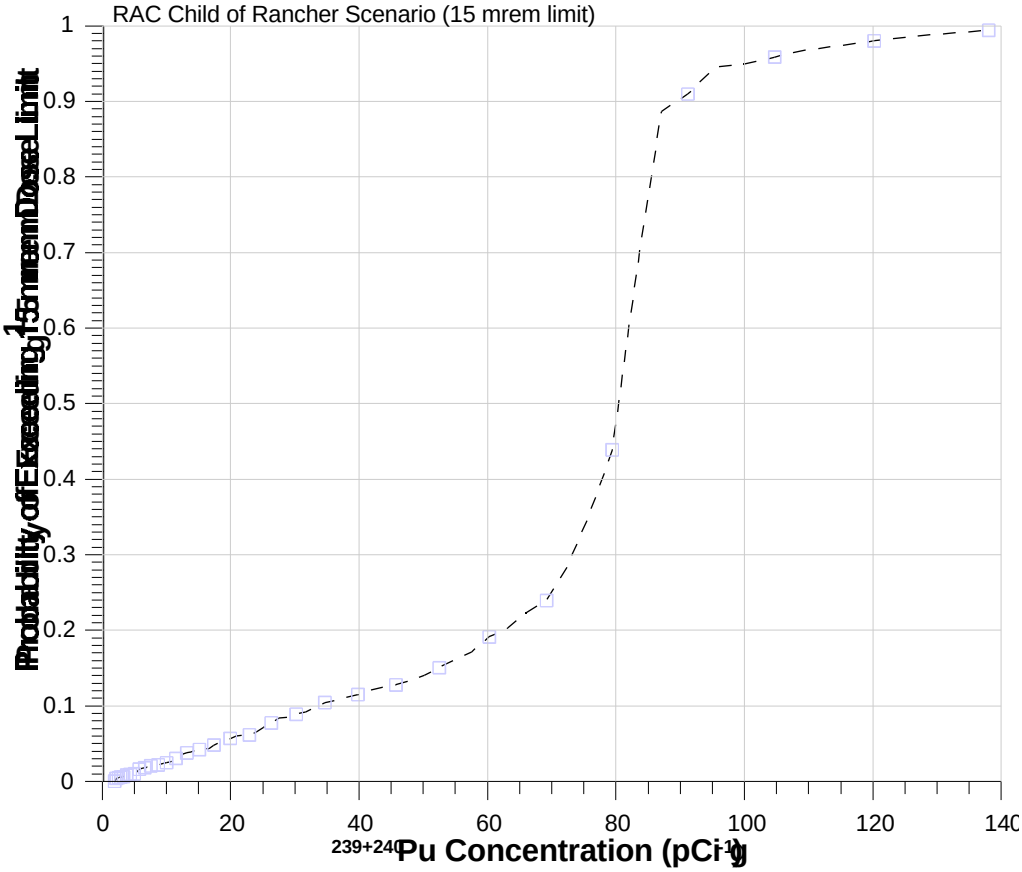

Figure 11-19. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

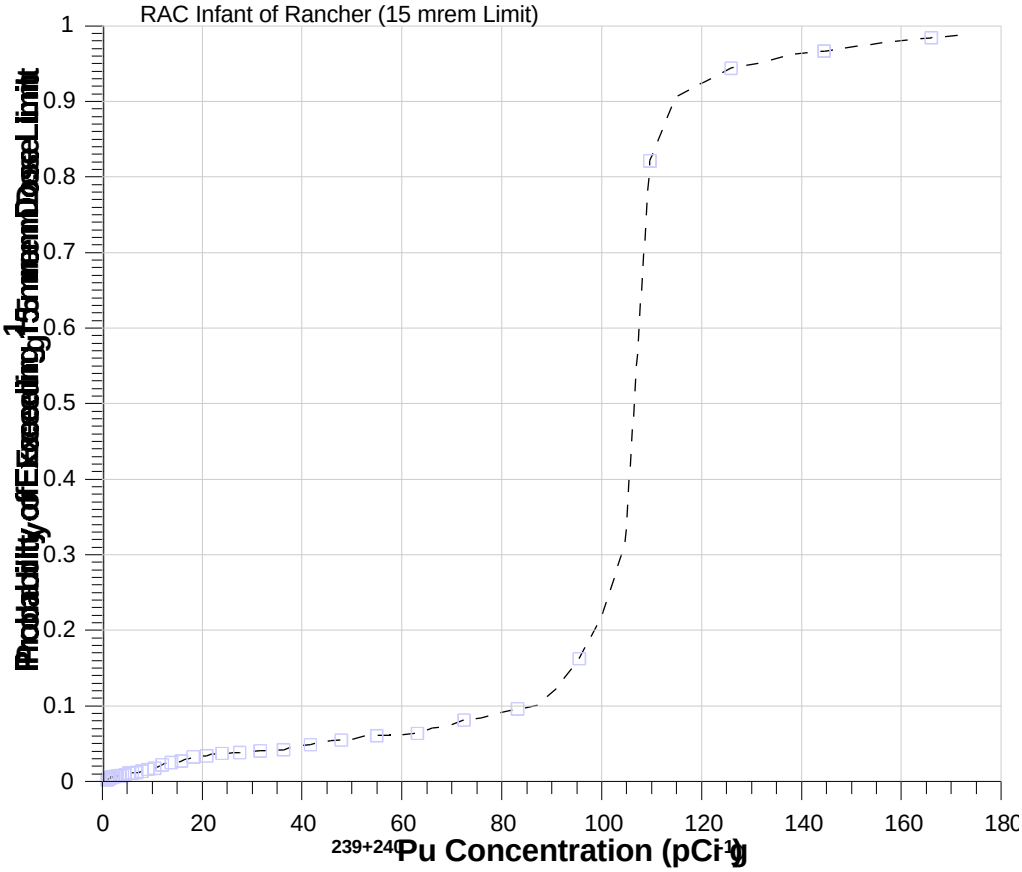

Figure 11-20. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* infant of rancher scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

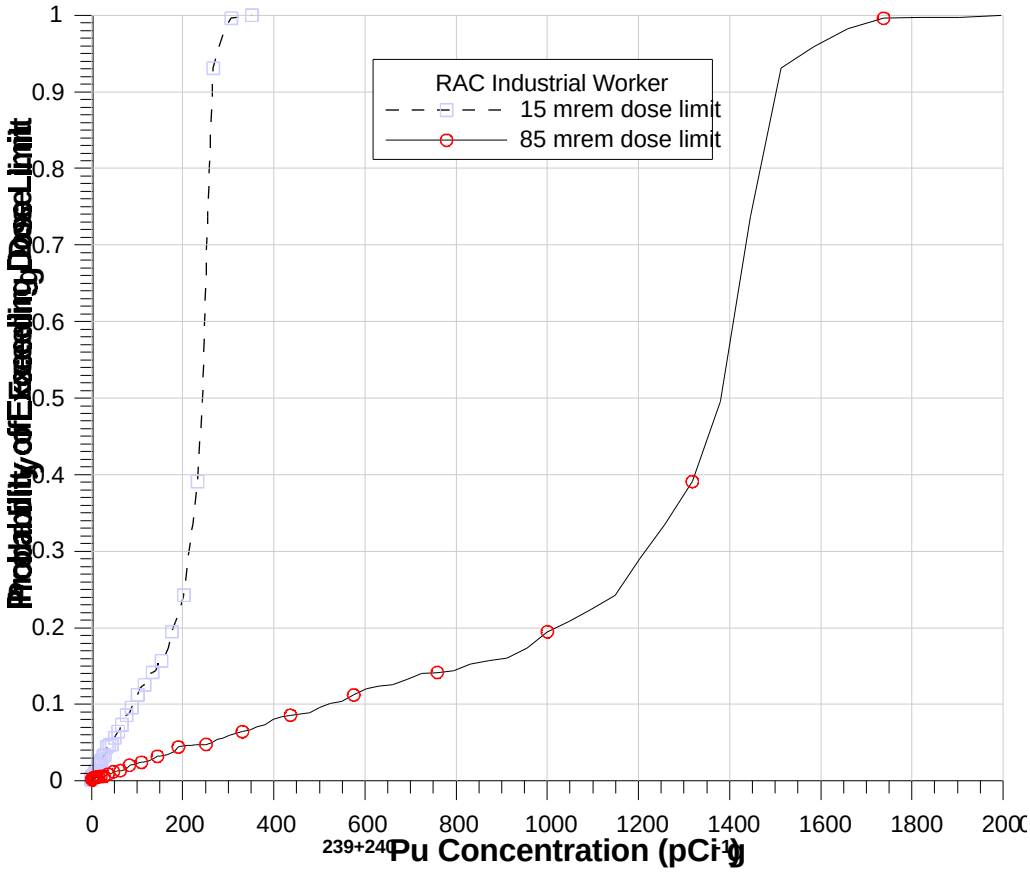

Figure 11-21. Probability of the total dose exceeding the 15 and 85 mrem dose limits for the *RAC* site industrial worker scenario. Total dose includes the sum-of-ratios calculation for all plutonium isotopes and their daughter products. This probability curve includes the impact of a fire considered probabilistically.

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:

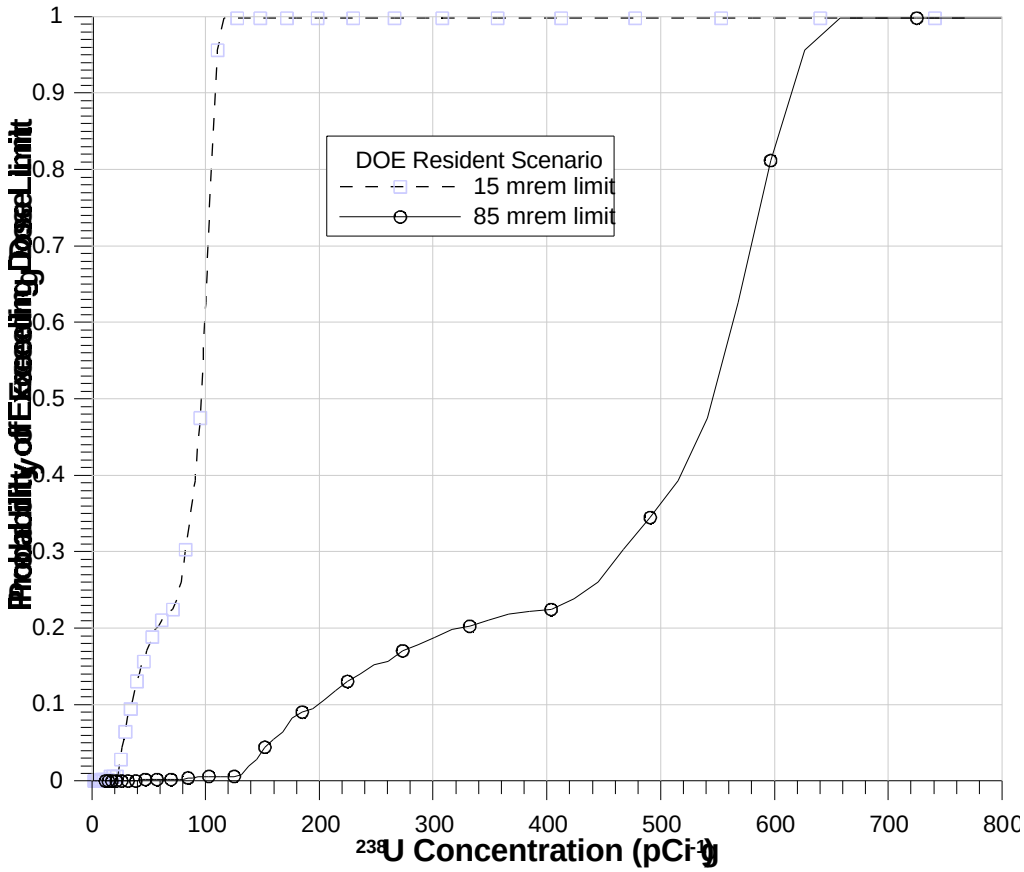

Figure 11-22. Probability of the total dose exceeding the dose limit for the DOE/EPA/CDPHE Resident scenario. Total dose includes all uranium isotopes and their daughter products.

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:

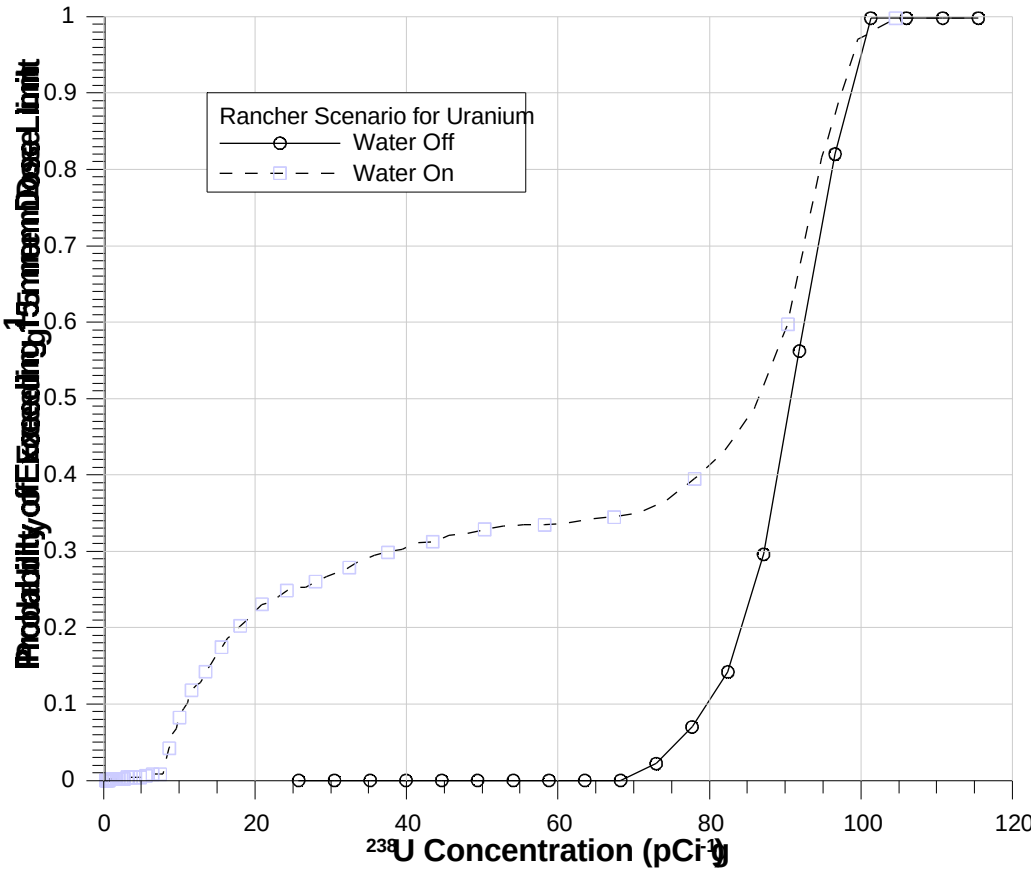

Figure 11-23. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* resident rancher scenario. Total dose includes all uranium isotopes and their daughter products.

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:

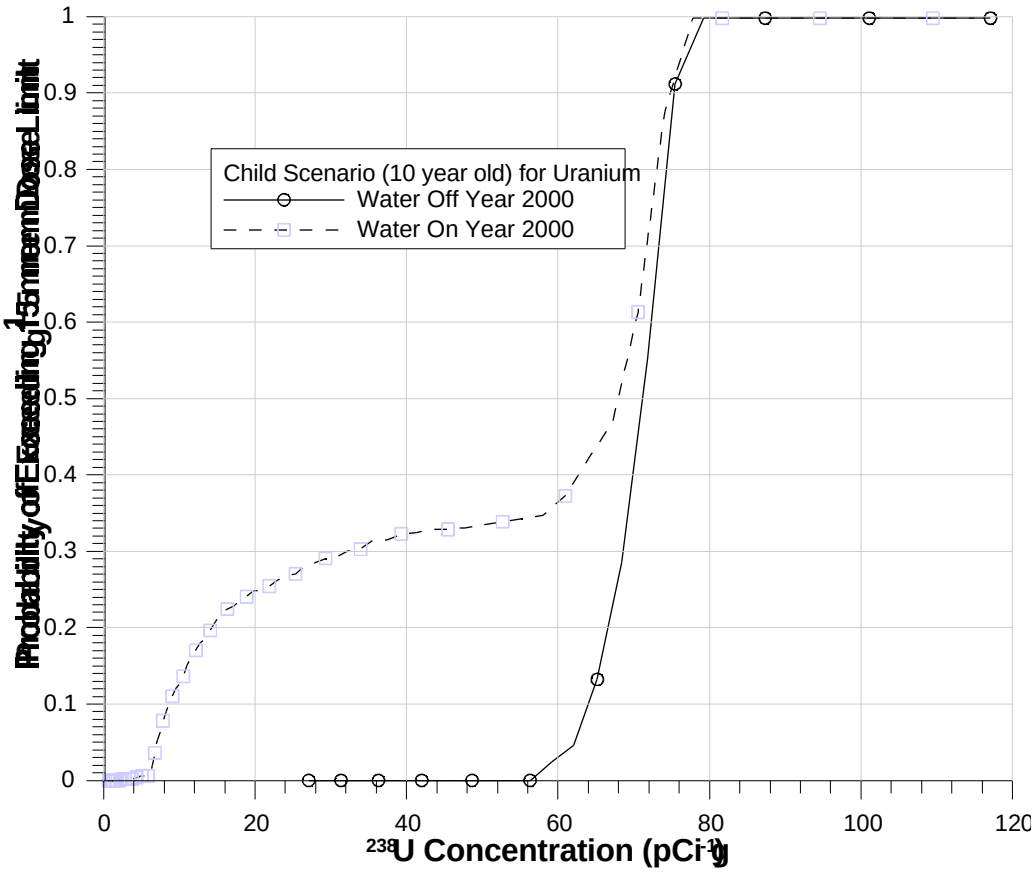

Figure 11-24. Probability of the total dose exceeding the 15 mrem dose limit for the *RAC* child of rancher scenario. Total dose includes all uranium isotopes and their daughter products.

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, February 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.

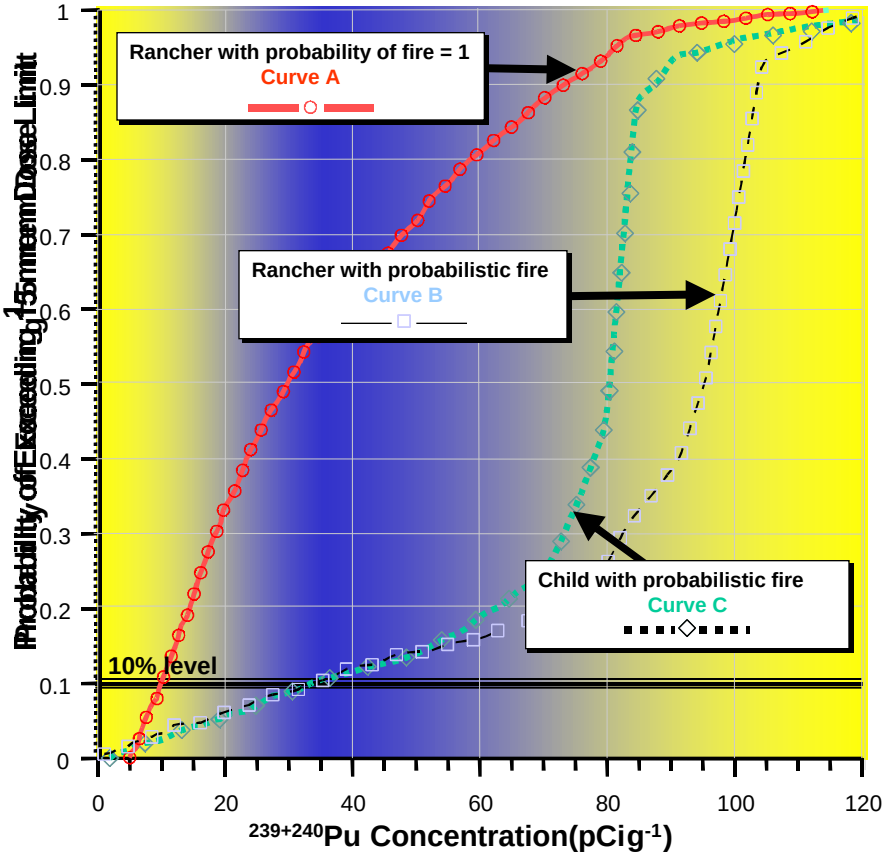

Figure 13-1. Composite graphic illustrating the most restrictive scenarios and showing a region centered at a soil action level of 35 pCi g^{-1}. Curve A represents the rancher and assumes that a fire occurs with a probability of 1; curve B represents the rancher scenario and takes into account the occurrence 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 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}.

14. REFERENCES

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APPENDIX A

COMPUTATIONAL DETAILS

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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
             # 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 $aa0, $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))
         [ "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);
    }
```

```
        $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.

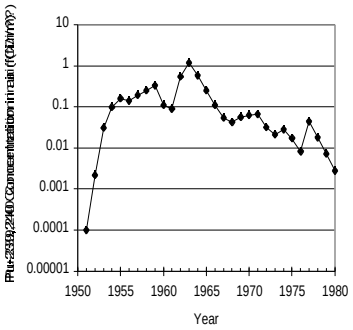

Figure H-1. Predicted surface air concentrations of ^{239,240}Pu (fCi m^{-3}) from global fallout from nuclear weapons testing. Predictions are for the mid-latitudes in the northern hemisphere.

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).

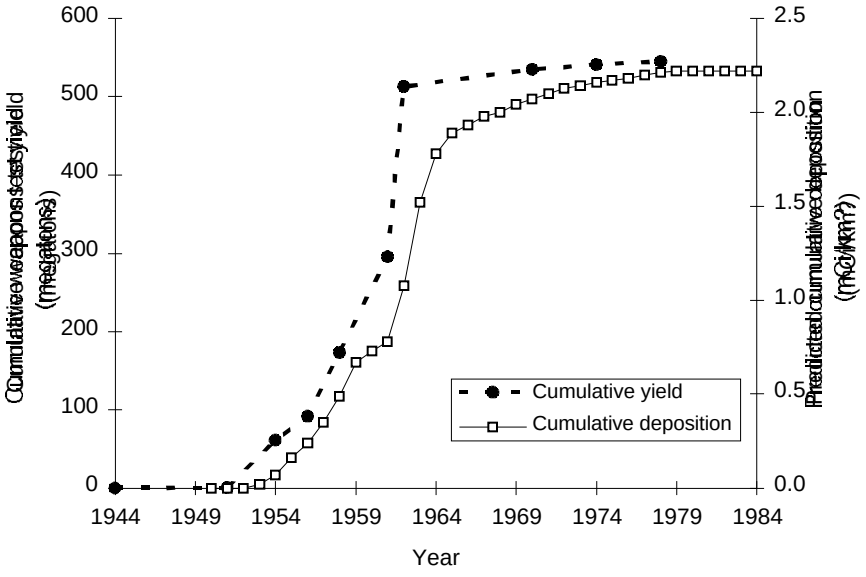

Figure H-2. Predicted cumulative deposition of plutonium in the New York area because of global fallout from nuclear weapons testing. For comparison, the cumulative explosive yield of atmospheric weapons tests is also plotted.

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.

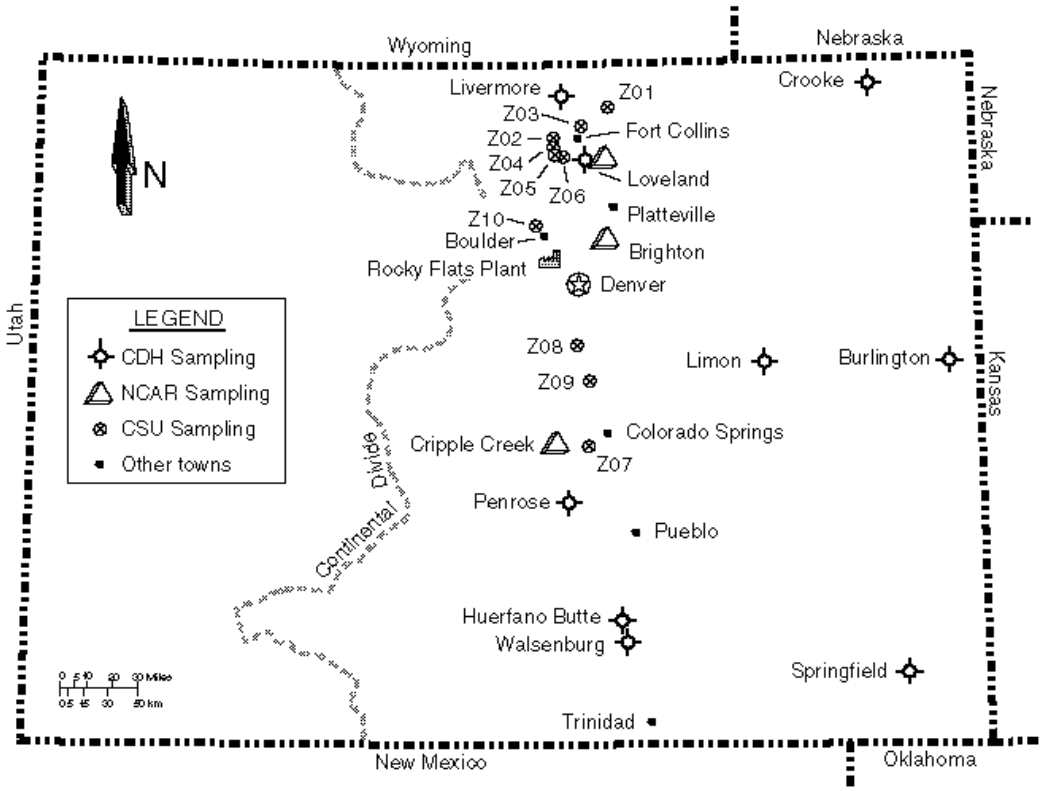

Figure H-3. Background sample locations in Colorado established by the National Center for Atmospheric Research, Colorado Department of Health, and Colorado State University soil studies (after the 1969 fire).

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.

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)

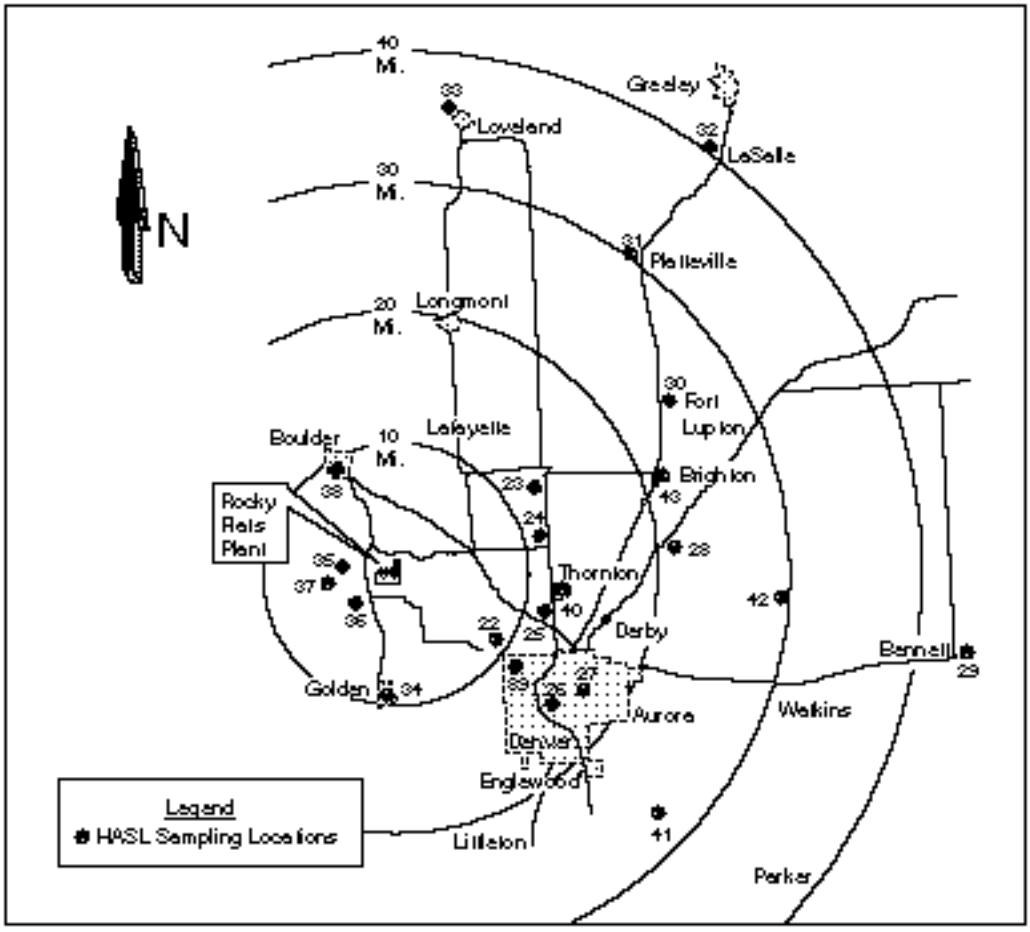

Figure H-4. Locations of the Health and Safety Laboratory soil sampling around the Rocky Flats Plant. Locations 1–33 were used in the 1970 sampling (Krey and Hardy 1970; Seed et al. 1971) and the 1971 sampling (Krey and Krajewski 1972). Locations 34–43 were added in the 1972 sampling (Krey 1976). Only 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

(±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 θ from the 903 Area (Bq m^{-2})

$\left[{}^{239}\mathrm{Pu}\right]^{\theta,D}_{0-3\,\mathrm{cm}}$ = concentration of plutonium in the 0 to 3-cm layer of soil at distance D and direction θ from the 903 Area (Bq kg^{-1}).

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.

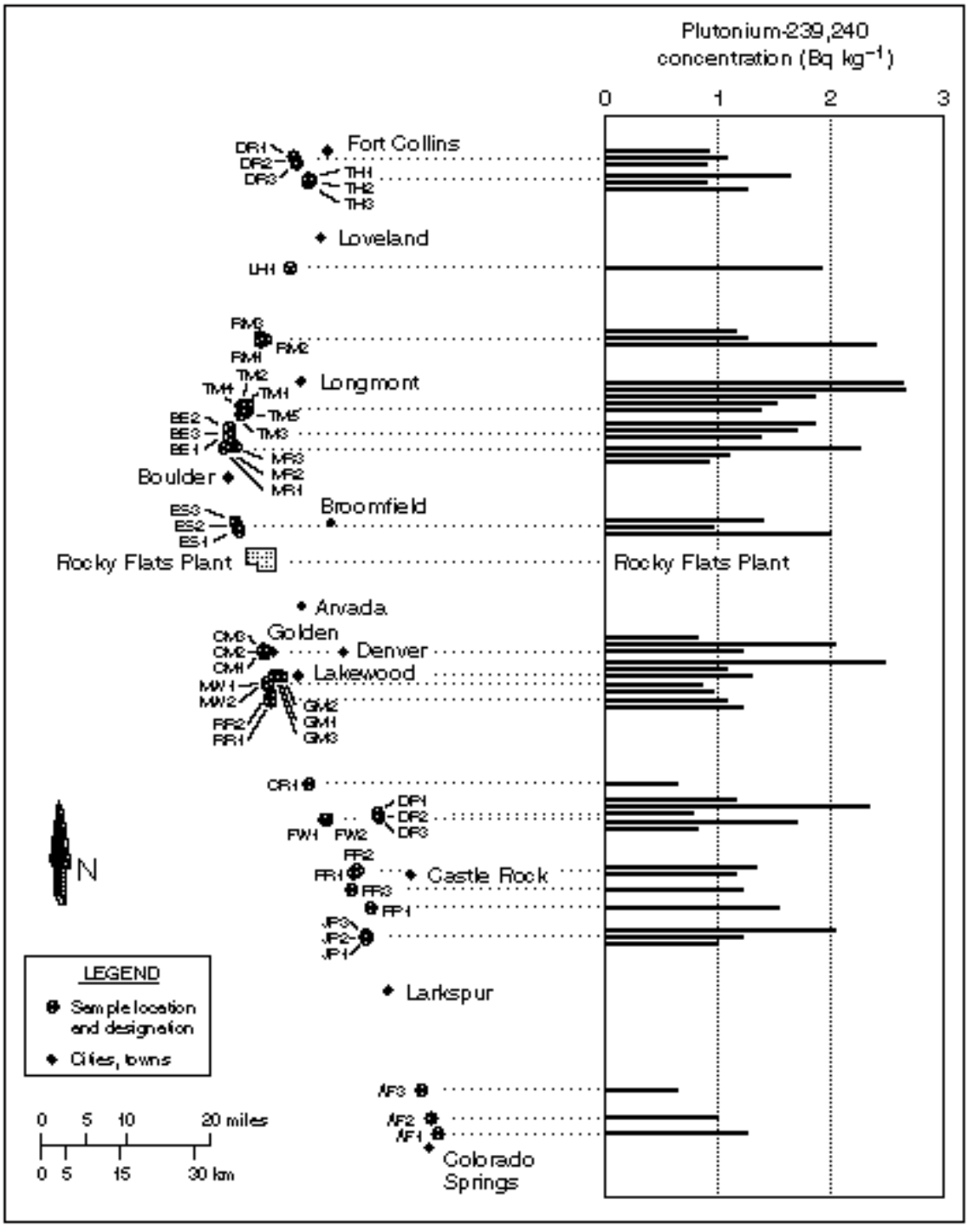

Figure H-5. Background sampling locations and concentrations of ^{239,240}Pu in surface soil along the front range, measured in 1994 (EG&G 1995). The left side of this figure shows the sampling locations, to scale. The right side shows the measured plutonium concentrations. For a given location, the measured concentration is shown directly to the right of the location. Data are from Table H-12.

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

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.

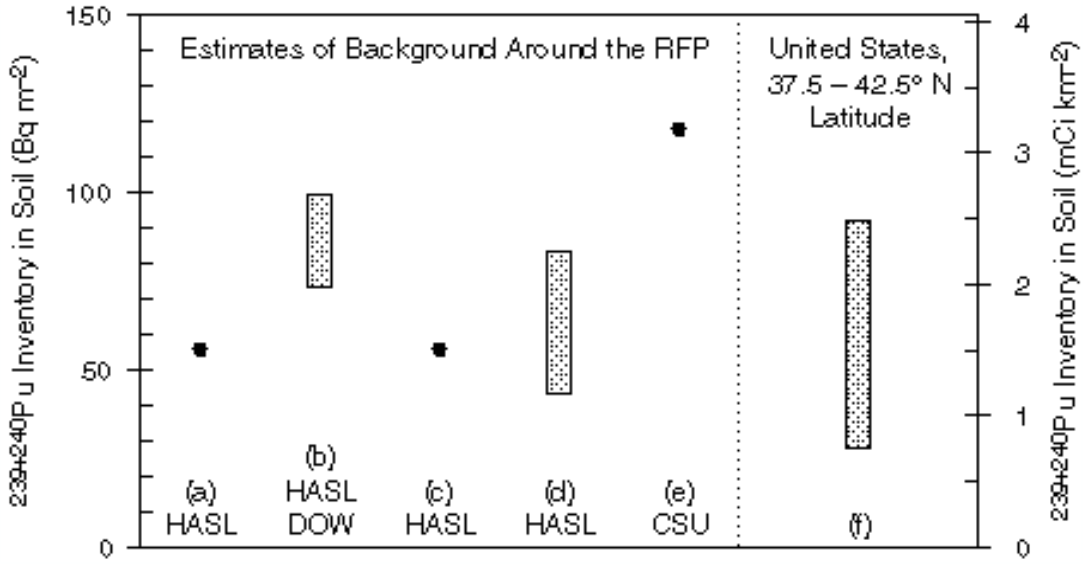

Figure H-6. Background ^{239,240}Pu total deposition (inventory) in soils: comparison of levels around the RFP with levels in the U.S. at similar latitude. Data are described in this appendix. Notes: (a) single location, in 1965, (b) examined distribution of measurements from 1970, (c) used ^{240}Pu/^{239}Pu 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 _ 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

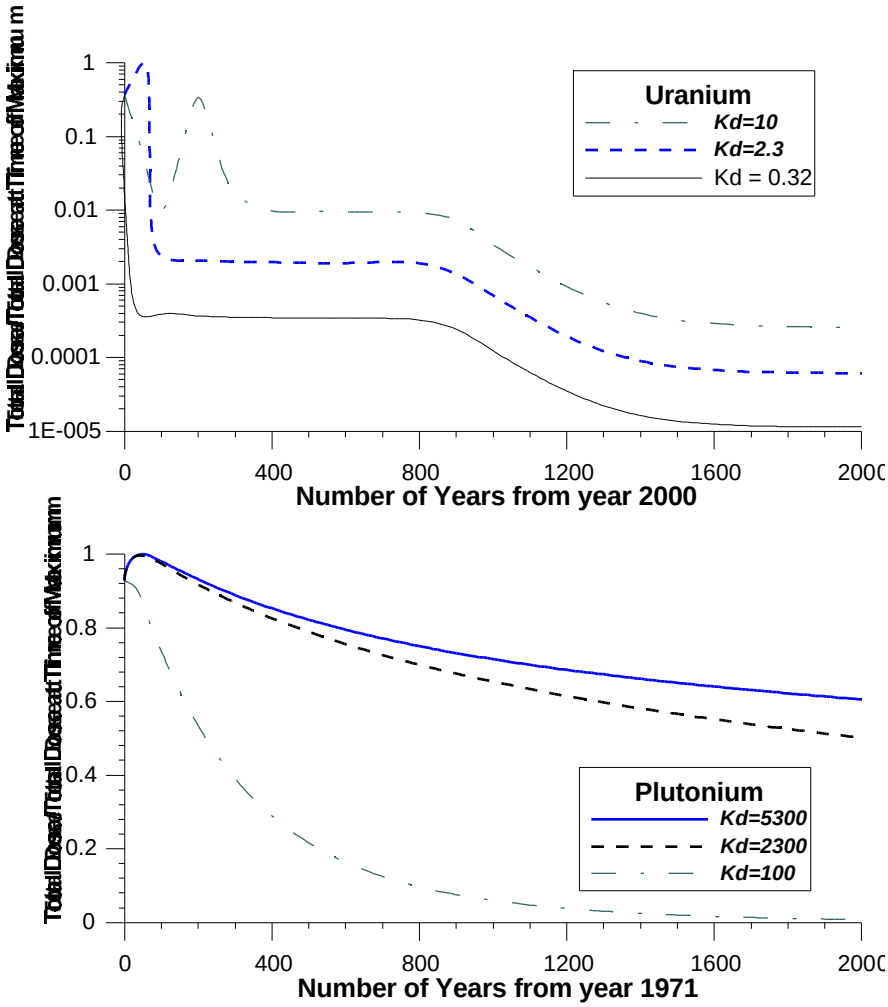

Figure D-1. The ratio of the total dose (all isotopes) for a given year to the maximum total dose for the simulation, as a function of time from the start of the simulation and for several different K_d values. Uranium isotopes include ^{234}U, ^{235}U, and ^{238}U. Plutonium isotopes include ^{238}Pu, ^{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 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[5]

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 (5^{th} percentile) below the nominal value of $5 \_ 10^{-2}$ Sv^{-1} to about twice the nominal value (95^{th} 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[6] 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 mrem y^{-1} $\quad$ $5 \_ 10^{-2}$ Sv^{-1} $\quad$ 10^{-2} Sv rem^{-1} $\quad$ 10^{-3} mrem rem^{-1} = $7.5 \_ 10^{-6}$ for each year [7].

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 (90^{th} 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 $9 \_ 10^{-4}$ (5^{th} to 95^{th} 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

[5] Adapted from material provided by W.K. Sinclair, Ph.D.

[6] 1 sievert (Sv) = 100 rem

[7] 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 $5 \leftrightarrow 10^{-2}$ Sv^{-1} because the nominal risk is for a population of all ages.

bone marrow, as can be seen from the data on dose per unit intake given in ICRP Publication 71 (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 _ 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 _ 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[8]. 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 _ 10^{-5} to about 7 _ 10^{-3}.

[8] 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.

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	$0.05 \_ 10^{-5}$		$\underline{\_\ 1.03}$ $= 71.4 \_ 10^{-8}\ Bq^{-1}$
Remainder	$3.4 \_ 10^{-7}$	0.05	$0.0017 \_ 10^{-5}$		or
Thyroid	$3.2 \_ 10^{-7}$	0.05	$0.0016 \_ 10^{-5}$		$= 7.11 \blacklozenge 10^{-6}\ Sv^{-1}$
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

sensitizers, inhibitors, smoking, alcohol, hormonal factors, etc., to be quantified. EPA, in many discussions on the subject, has made a target based on risk; namely that a lifetime risk in the range of 10^{-4} to 10^{-6} should be aimed at as a limiting factor in regulatory decisions (EPA 1996, p 19)[9]. Presumably, a lifetime risk of 10^{-4} is not to be exceeded and 10^{-6} is the lowest risk worth noting (i.e., risks below 10^{-6} are negligible).

Nevertheless, risk *limits* have apparently never been specified as such and in the case of contaminated soil, EPA finds the lifetime risk ($3 \_ 10^{-4}$) associated with 15 mrem y^{-1} "protective" and acceptable as a limit (EPA 1996, p15) even though it exceeds the target. However, the risk associated with 25 mrem y^{-1} ($5 \_ 10^{-4}$) is apparently not sufficiently protective of the public (EPA 1996, p 34). Thus, there is an arbitrariness about just where to draw the line, an arbitrariness that unfortunately dogs all regulatory decisions of this type. Another well-known circumstance at variance with the target risk is EPA's guidance level for domestic indoor radon that is set at advising remediation for radon levels at 4 pCi L^{-1} or more (EPA and DHHS 1986). This corresponds to a lifetime risk of $1–2 \_ 10^{-2}$, well above EPA's normal aim for risk control (see footnote e).

While a specified risk limit may be attractive for all carcinogens, in the case of radiation, a specified risk limit is at least superficially less satisfactory than a specified dose limit, for two reasons. First, dose is a physical measure of the amount of radiation in question and can readily be measured, which is a very important consideration when establishing limits. Second, estimates of risk per unit dose have been changing substantially over time and have only recently appeared to stabilize (UNSCEAR 2000). Unfortunately, risk, especially the type of latent risk resulting from carcinogenesis, is not readily measurable and must be calculated often from long-term studies of the effects of the agent in question on populations. Inherent in many of these studies are uncertainties in the estimates of risk, some of which cannot easily be reduced. Consequently, at the present time, it would seem more appropriate to do as DOE/EPA/CDPHE (1996) have done and firmly specify a dose limit for the maximally exposed individual but to note the approximate risk that is associated with this limit as discussed in Section E.2. It is this effective dose limit[10] (15 mrem y^{-1} for 30 years) that has formed the basis of the risk estimates calculated in this appendix. These are maximum risk estimates. The RSALs calculated in the main body of this report are based on the 15 mrem y^{-1} being reached only once as the result of a fire, and all other years were assumed to be at a lower exposure. Thus, the risk could range from $7.5 \_ 10^{-6}$ as a minimum (1 year at 15 mrem y^{-1} with all other years at zero exposure) to $2.3 \_ 10^{-4}$ as a maximum (30 years at 15 mrem y^{-1}). Each of these limits is unlikely to represent the actual risk; rather, the risk falls somewhere between them.

[9] In a preamble to a recently proposed rule for radionuclide MCLs, the Agency stated that "longstanding and carefully considered EPA policy for regulating carcinogens in drinking water is that the lifetime individual risk *target* is 1 in 10,000 (10^{-4}) to 1 in 1,000,000 (10^{-6}) risk {56FR33058}, cited in EPA (1996).

[10] In EPA (1996) (for example p.9, p.28) the term "effective dose equivalent" is used, but this has been replaced by "effective dose" with different component weighting factors and different estimates of organ risk by ICRP in 1991. It is not clear whether both the organ risks and the weighting factors have been updated.

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