RSALOP

Radionuclide Soil Action Levels Oversight Panel

Task Reports 1, 2, 3
and
Comments

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DRAFT REPORT

Task 1: Cleanup Levels at Other Sites

Rocky Flats Citizens Advisory Board
Rocky Flats Soil Action Level Oversight Panel

February 1999

Submitted to the Rocky Flats Citizens Advisory Board
in Partial Fulfillment of Contract between RAC and RFCAB

"Setting the standard in environmental health"

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2

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Task 1: Cleanup Levels at Other Sites

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February 1999

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3

CONTENTS

TASK 1: CLEANUP LEVELS AT OTHER SITES

INTRODUCTION

The soil action levels for radionuclides calculated for the Rocky Flats Environmental Technology Site (RFETS) by the U.S. Department of Energy (DOE), U.S. Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE) have come under scrutiny because of lack of public involvement throughout their development. A soil action level is calculated to identify the concentration of radionuclides in the soil above which action should be taken to prevent people from receiving unacceptable radiation dose levels. As a result of public concern, DOE provided funds to the Rocky Flats Citizen's Advisory Board to establish the Rocky Flats Soil Action Level Oversight Panel and to hire a contractor to conduct an independent assessment and calculate soil action levels for Rocky Flats. *Risk Assessment Corporation (RAC)* was hired to perform the study.

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

ROCKY FLATS SOIL ACTION LEVEL CALCULATION

A 1996 report documents the original calculation of soil action levels for the RFETS (DOE 1996). The RESRAD computer code (Yu et al. 1993) was used, and action levels were calculated for three different land use scenarios at two different effective dose equivalent levels.

The three scenarios established for Rocky Flats were (1) an open space exposure scenario that assumed no development in the area, (2) an office worker exposure scenario, and (3) a hypothetical future resident scenario. Action levels were calculated for ^{241}Am, ^{238}Pu, ^{239,240}Pu, ^{241}Pu, ^{242}Pu, ^{234}U, ^{235}U, and ^{238}U. Public concern has been the highest for the ^{239,240}Pu action level; therefore, we focused our efforts on the this action level during the Task 1 study.

The open space and office worker scenarios were based on the principle that the land currently occupied by the RFETS will remain under institutional control for 1000 years. Under institutional control, no person would be allowed to live on current site property; however, the site would be occupied by office buildings and open recreational space. If institutional control failed, anything could happen to the land, and the scenarios with the largest potential exposure would be assumed to occur. This large exposure is represented by the hypothetical future resident scenario, which describes a resident who lives full-time on the former site, farming and eating crops grown on the land.

The dose levels that drive the calculations of action levels for the scenarios are annual effective dose equivalents of 15 and 85 mrem, depending on the scenario and the status of the institutional controls. These dose levels were selected based upon combined regulatory guidance from the EPA and DOE and are presumed to be protective of human health (DOE 1996).

The Task 1 study uses the hypothetical future resident 85 mrem y^{-1} action level because it is the DOE recommended action level above which no remediation would be required, and it is the most readily comparable action level to those at other facilities. This report uses the 85 mrem y^{-1} action level to make all comparisons.

Pathways Considered

The original RFETS calculation, documented in DOE (1996), established a site conceptual model based on the environment at Rocky Flats. Pathway analyses were performed based upon this model. This analysis allowed DOE to select the appropriate pathways in RESRAD for use in the RFETS soil action level determination. Potential pathways available in RESRAD are:

- External gamma exposure
- Soil inhalation
- Plant ingestion
- Meat ingestion
- Aquatic food ingestion
- Groundwater and surface water ingestion
- Soil ingestion
- Radon exposure.

Of these pathways, only external gamma exposure, soil inhalation, plant ingestion, and soil ingestion were assessed for the hypothetical future resident. As described in DOE (1996), the other pathways were eliminated from consideration because of inconsistencies with the site conceptual model, absence of pathways within the Rocky Flats environment, or insignificant contribution to the total dose. For example, aquatic food ingestion is not consistent with the site conceptual model because there are no surface water sources on the site that can sustain a fish population (DOE 1996). Differences in pathways analyses among the sites compared in this paper are noted in the following paragraphs.

Important Parameters

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

METHOD OF COMPARISON

Action and cleanup levels are sometimes determined independent of dose levels or are based on a different dose than the 85 mrem y^{-1} used in the RFETS hypothetical future resident scenario calculation. This fact makes direct comparison more difficult; therefore, we compared different soil action levels among sites by normalizing the action level to annual dose. In the remainder of this paper, annual dose is understood, and dose is represented in units of millirem. Normalization means that a ratio was calculated for action level to dose level, representing the action level for a unit dose, or 1 mrem. This equitable comparison allows for straightforward identification of pathway, scenario, and parameter differences that affect the ratio. If these differences can be identified among the RFETS and other sites, the ratios between sites should be comparable.

Each ratio is identified in two ways:

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

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

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

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

SOIL ACTION LEVELS AT OTHER SITES

We identified several sites and alternate action level calculations for comparison in the Task 1 report. These included

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

- Palomares, Spain.

Table 1 identifies the dose to soil action level and soil action level to dose ratios for each site where information was available. All ratios are shown for $^{239,240}Pu$ unless otherwise indicated. The ratio for the most comparable scenario to the RFETS residential scenario is shown for each site. In each case, this is a residential scenario where remediated land would be lived on and, in some cases, farmed. Ratios and scenarios are described in more detail in the following sections.

Table 1. Ratios for Comparison among Different Sites[a]

Site	Soil action level to dose ratio ($[pCi\ g^{-1}]\ mrem^{-1}$)	Dose to soil action level ratio ($mrem\ [pCi\ g^{-1}]^{-1}$)
Rocky Flats, Colorado	17	0.06
Hanford, Washington	2.3	0.44
Nevada Test Site[b]	4.1	0.24
NRC remediation codes	7.4	0.14
Johnston Atoll[c]	0.85	1.2
Maralinga, Australia	0.56	1.8
Semipalatinsk Nuclear Range	8.8	0.11
Palomares, Spain	12.3	0.08

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

It is clear that the values are not the same for all sites. In fact, the soil action level to dose ratio is less than 1 in some cases. We will now step through a site-by-site analysis of each ratio and why it differs from the ratio for the RFETS hypothetical future resident.

Hanford, Washington

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

The soil action level to dose ratio at Hanford is 2.3, over 7 times smaller than the same ratio at Rocky Flats. This ratio is for the Hanford rural residential scenario. This scenario represents a person who lives on the current Hanford site all year, eating crops and livestock grown onsite, drinking from site streams, inhaling air and ingesting soil. Hanford soil action levels were calculated using the RESRAD computer code.

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

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

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

A major difference between the Hanford and RFETS calculation is that plutonium at the Hanford reservation is assumed to be in a soluble form in the environment. Because of this assumption, the dose conversion factors used in the Hanford calculation are larger than those used in the RFETS calculation, where plutonium is assumed to be insoluble. Maintaining our previous pathway modification and now assuming the plutonium at Hanford is in an insoluble form like RFETS plutonium, the soil action level to dose ratio for $^{239,240}Pu$ changes from 2.3 to 9.9. This ratio is much closer to the RFETS ratio of 17, indicating that the form of plutonium identified in the environment plays a significant role in the difference between these two calculations.

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

When the Hanford calculations using RESRAD are run implementing the RFETS pathways and parameter values for mass loading and dose conversion factor, the soil action level to dose ratio for Hanford exceeds that for the RFETS. Table 2 shows the incremental change in the soil action level to dose ratio when the parameters in the Hanford calculation were altered.

Table 2. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Hanford and RFETS Calculations

Location	Parameter change	Soil action level to dose ratio ([pCi g^{-1}] mrem^{-1})	Dose to soil action level ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats	Original calculation	17	0.06
Hanford	Original calculation	2.3	0.44
	Remove meat, milk, fish, drinking water	2.3	0.44
	+ change dose conversion factor	9.9	0.10
	+ change mass loading	34	0.03

Nevada Test Site

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

Another document assessed dose for presumed cleanup levels given scenarios similar to those we have looked at for the RFETS (DOE-NV 1997). This assessment was performed with RESRAD but in reverse to the RFETS calculations.

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

The rancher scenario resulted in the maximum dose for the same soil concentrations. In this scenario, a person lives on and farms the land for personal livelihood, eating many of the crops and livestock produced. For a soil concentration before remediation of 326 pCi g^{-1}, for Clean Slate Site 1, the corresponding dose was 78.3 mrem y^{-1}. The soil action level to dose ratio for this facility was 4.2 (pCi g^{-1}) mrem^{-1}. The same ratio applied to the post-remediation soil concentration level of 162 pCi g^{-1} and dose of 38.9 mrem y^{-1}.

The primary difference between the RESRAD calculations for the NTS and the RFETS is the assumed solubility class of plutonium. The NTS calculation used the RESRAD default value for plutonium dose conversion factor, which corresponds to Class W (soluble) plutonium. When dose conversion factors for soluble plutonium are used in the Rocky Flats calculation, which originally used Class Y (insoluble) plutonium dose conversion factors, the soil action level changes from 1429 to 242 pCi g^{-1}, and the soil action level to dose ratio changes from 17 to 2.8 (pCi g^{-1}) mrem^{-1}. This single parameter accounts for the difference between these two calculations. Table 3 summarizes the differences between the ratios and the parameter changes employed.

10 **DRAFT**

Table 3. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for the NTS and RFETS Calculations

Location	Parameter change	Soil action level to dose ratio ($[pCi\ g^{-1}]\ mrem^{-1}$)	Dose to soil action level ratio ($mrem\ [pCi\ g^{-1}]^{-1}$)
Rocky Flats	Original calculation	17	0.06
	Change dose conversion factor	2.8	0.36
Nevada Test Site	Original calculation	4.1	0.24

U.S. Nuclear Regulatory Commission DandD Code Scenarios

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

This scenario assumes residential use of land with limited gardening activities. The three major pathways considered are inhalation, ingestion of food products grown in contaminated soil, and external gamma exposure. Indoor radon is not considered. Of particular interest in the DandD code is the distinction between time spent indoors, outdoors, and outdoors gardening and the different mass loading factors applied to each time period. All NRC mass loading factors are larger than the RFETS mass loading factor of 0.000026 g m^{-3}.

The total effective dose equivalent for the residential scenario for $^{239,240}Pu$, assuming surface soil activity of 1 pCi g^{-1}, is 0.14 mrem. This gives a dose to soil action level ratio of 0.14 mrem $(pCi\ g^{-1})^{-1}$ and a soil action level to dose ratio of 7.1 $(pCi\ g^{-1})\ mrem^{-1}$ (NRC 1990).

The dose conversion factor used for inhalation is the same as that used for the RFETS calculation, so we might assume that the difference in the value of the mass loading factor causes the difference between the NRC and RFETS ratios. To explore this possibility, we used the Rocky Flats RESRAD calculation and input NRC mass loading factors.

The three mass loading factors used in the NRC calculation are for indoor mass loading, outdoor mass loading, and outdoor mass loading during gardening activities. Because the RFETS RESRAD calculation assumes indoor air concentration is equal to outdoor air concentration and gardening activities are not included, we used the NRC outdoor mass loading factor of 0.0001 g m^{-3} to input into the RFETS calculation. This mass loading factor changed the soil action level to dose ratio for ^{239}Pu from 17 to 4.6 $(pCi\ g^{-1})\ mrem^{-1}$.

The single change in magnitude of mass loading made the adapted RFETS soil action level to dose ratio (4.6) smaller than the same NRC ratio (7.1), indicating less allowable soil concentration for the same dose.

In the DandD code, the dose conversion factors are maximized for each intake pathway. That is, for soil ingestion, soluble plutonium dose conversion factors are used, and for inhalation, insoluble dose conversion factors are used. Using different dose conversion factors maximizes the dose and minimizes the acceptable soil action level. Overall, the NRC code appears to be very

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conservative, and the parameter values for each scenario were chosen to promote conservatism. If certain parameters about the site are not known, these conservative values can be used as defaults. Within the text of the NRC reports discussing this code, however, it is cautioned that if site-specific values are available, they should be used to provide a more realistic assessment of the cleanup needs (NRC 1990).

Table 4 summarizes the ratios for the NRC DandD code and the RFETS calculations, and it documents the changes made to account for the differences between the values.

Table 4. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for NRC DandD and RFETS Calculations

Location	Parameter change	Soil action level to dose ratio ($[pCi\ g^{-1}]\ mrem^{-1}$)	Dose to soil action level ratio ($mrem\ [pCi\ g^{-1}]^{-1}$)
Rocky Flats	Original calculation	17	0.06
	Change mass loading	4.6	0.22
NRC DandD Code	Original calculation	7.1	0.14

Johnston Atoll, Marshall Islands

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

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

$$SSL = \frac{C_{air,acceptable}}{ML \cdot EF} \qquad (1)$$

where

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

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

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

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

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

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

The first step was to determine the difference between dose conversion factors for the two sites. To extract the Johnston Atoll dose conversion factor from the existing information, we used an equation for effective dose from inhaled material. Equation (2) calculates dose (in units of millirem) from inhaled material.

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

where

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

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

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

Equation (1) is used to calculate the Johnston Atoll soil screening limit using Rocky Flats values. The Rocky Flats value for mass loading was 0.000026 g m^{-3}. The air concentration was calculated above, and in the RFETS calculation, no enrichment factor was employed. The soil screening limit for Johnston Atoll using RFETS parameter values is 356 pCi g^{-1}, giving a soil action level to dose ratio of 17.8 (pCi g^{-1}) $mrem^{-1}$, which matches that of the RFETS. Table 5 summarizes the results of this analysis.

Table 5. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Johnston Atoll and RFETS Calculations

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

Enewetak Atoll, Marshall Islands

The cleanup levels established for the Enewetak Atoll are difficult to compare to the Rocky Flats soil action levels. This cleanup was driven more by time, money, and military concerns than an identified limit for concentrations in soil.

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

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

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

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

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

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

The Bair Committee approved of the ERDA Operating Plan cleanup criteria and suggested that more specific guidance be established for the soil concentrations between 40 and 400 pCi g^{-1}. When the 1977 EPA guidance on transuranics was released, the Bair Committee adapted its recommendations for agricultural land soil concentrations to 80 pCi g^{-1} and food gathering land soil concentrations to 160 pCi g^{-1}. These values were apparently based on a dose assessment study performed by Lawrence Livermore Laboratory. A first study done by Lawrence Livermore Laboratory was based on the original soil cleanup criteria, but the results were deemed incorrect because of a mathematical error. The Laboratory performed a new dose assessment. Results from this new dose assessment influenced the Bair Committee's decisions concerning action levels for different soil uses.

We could not locate the Lawrence Livermore Laboratory study in the literature. The Defense Nuclear Agency document lists the doses from this study only in radiation doses in millirad; however, these values cannot be converted to effective doses without knowing more about the dose model used to make the calculations. We can assume that Lawrence Livermore Laboratory scientists used the same model as that used in the Johnston Atoll study, with a large dose conversion factor. However, we would need to have access to the Lawrence Livermore Laboratory study to make comparisons to RFETS values.

Maralinga, Australia

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

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

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

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

where

C_{air} = concentration in air (pCi m^{-3})

BR = breathing rate (m^3 y^{-1})

DCF = dose conversion factor (mrem pCi^{-1})

and

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

where

C_{soil} = soil concentration (pCi g^{-1})

ML = mass loading (g m^{-3}).

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

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

where all quantities are as previously defined.

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

Mass loading for the site was determined by simulating some Aboriginal dust raising activities. These data were the only data available to the Australian Radiation Laboratory group, and a value of 0.001 g m^{-3} was used for adults. Breathing rates were taken by the Australian

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

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

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

Using the Rocky Flats values in Equation (5) accounts for the difference in the two ratios. Table 6 summarizes the changes in the ratios between Maralinga and the RFETS by altering the parameter values used in the calculation.

Table 6. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Maralinga and RFETS Calculations

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

Semipalatinsk Nuclear Range, Kazakhstan

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

It is important to point out that the values given in the literature usually document either a range of surface radiation levels associated with a single dose or a range of doses associated with a single radiation level. It is very difficult to correlate dose to corresponding soil concentration, but this paper presents the best ratios we could determine. Zeevaert et al. (1997) should be carefully reviewed if more information is desired.

For settlements at the Semipalatinsk site, maximum soil activity was given as 11 kBq m^{-2}, corresponding to a soil concentration of 1.32×10^6 pCi g^{-1}. We assumed a depth of contamination of 15 cm and a soil density of 1.5 g m^{-3} because these factors were not given in Zeevaert et al. (1997). The dose resulting from this concentration is identified as 1.5 Sv, or 150,000 mrem. It is not clear that this dose is due to inhalation of contamination because it is identified only as the estimated individual dose to the population.

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

Another territory affected by the Semipalatinsk tests was Ouglovski, with soil concentrations of 6.6×10^5 pCi g^{-1}. The doses cited for this region are external doses, however, and cannot be applied to obtain a ratio.

Table 7 outlines the differences between Rocky Flats and the Semipalatinsk Nuclear Range. It is important to remember the differences in the source of these values. They are presented here in an attempt to make this review as complete as possible.

Table 7. Soil Concentration to Dose Ratio for $^{239,240}Pu$ for Semipalatinsk Nuclear Range Measurements and RFETS Calculations

Location	Soil action level to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action level ratio (mrem [pCi $g^{-1}]^{-1}$)
Rocky Flats	17	0.06
Semipalatinsk Nuclear Range	8.8	0.11

Thule, Greenland

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

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

Palomares, Spain

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

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

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

Air concentrations measured in the environment were compared to (a) annual limits on intake and (b) derived air concentrations from these limits as recommended by the ICRP for radiation workers (ICRP 1978). Because values for acceptable air concentrations for the public were not provided in ICRP (1978), the radiation worker values were multiplied by the ratio of dose limits recommended for the public to those recommended for radiation workers (0.1). This concentration was again reduced to account for ICRP recommendations that effective dose equivalent throughout the life of a member of the exposed population does not exceed the value resulting from a 1 mSv (100 mrem) annual effective dose equivalent. Therefore, acceptable concentration values for members of the public were set at 1.2 mBq m^{-3} (3.2×10^{-2} pCi m^{-3}) for Class Y (insoluble) compounds of plutonium and 0.5 mBq m^{-3} (1.35×10^{-2} pCi m^{-3}) for Class W (soluble) compounds of plutonium. In the context of the RFETS parameter values, with insoluble Class Y plutonium and a mass loading factor of 0.000026 g m^{-3}, this air concentration corresponds to a soil concentration of 1230 pCi g^{-1}.

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

Placing the breathing rate of 8395 m^3 y^{-1} into the RFETS calculation yields a soil action level of 1202 pCi g^{-1} and a soil action level to dose ratio of 14.1 (pCi g^{-1}) mrem^{-1}. We did not discover the reason for the remaining difference between these two ratios during this assessment. We have requested additional documents and we will complete a further analysis before the final draft of this paper is prepared in an attempt to identify the parameter(s) that accounts for the remaining difference.

Table 8 summarizes the changes made to the RFETS calculation and ratio.

Table 8. Soil Action Level to Dose Ratio for $^{239,240}Pu$ Changes with Parameter Alteration for Palomares and RFETS Calculations

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

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

CONCLUSIONS

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

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

Breathing rate is less significant because the range of possible values is limited to within reasonable boundaries. The dose conversion factor varies depending on the assumed solubility of plutonium. For soluble Class W plutonium, the inhalation dose conversion factor is 0.429 mrem pCi^{-1} and the ingestion dose conversion factor is 0.0035 mrem pCi^{-1}. For insoluble Class Y plutonium, the inhalation dose conversion factor is 0.308 mrem pCi^{-1} and the ingestion dose conversion factor is 0.000052 mrem pCi^{-1} (ICRP 1978). When soluble plutonium is assumed, the ingestion pathway dominates dose and the dose per unit intake is much greater. For the RFETS, we can determine the appropriate assumption based upon the oxidation state of the plutonium found in the soil at Rocky Flats.

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

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

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

Table 9. Summary of Comparisons between RFETS Calculations and Those for Other Facilities

Location	Parameter change	Soil action limit to dose ratio ([pCi g^{-1}] $mrem^{-1}$)	Dose to soil action limit ratio (mrem [pCi g^{-1}]$^{-1}$)
Rocky Flats	Original calculation	17	0.06
Hanford	Original calculation	2.3	0.44
	Remove meat, milk, fish, and drinking water pathways and change to RFETS dose conversion factor and mass loading	34	0.03
Rocky Flats	Original calculation	17	0.06
	Change to NTS dose conversion factor	2.8	0.36
Nevada Test Site	Original calculation	4.1	0.24
Rocky Flats	Original calculation	17	0.06
	Change to NRC mass loading	4.6	0.22
NRC DandD Code	Original calculation	7.1	0.14
Rocky Flats	Original calculation	17	0.06
Johnston Atoll	Original calculation	0.85	1.2
	Change to RFETS mass loading, enrichment factor, and calculate air concentration using RFETS dose conversion factor and breathing rate	17.8	0.056
Rocky Flats	Original calculation	17	0.06
Maralinga	Original calculation	0.56	1.8
	Change to RFETS mass loading, breathing rate, dose conversion factor	17.8	0.056
Rocky Flats	Original calculation	17	0.06
Semipalatinsk Nuclear Range	Original measurement	8.8	0.11
Rocky Flats	Original calculation	17	0.06
	Change to Palomares breathing rate	14.1	0.07
Palomares	Original calculation	12.3	0.08

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DRAFT FINAL REPORT

Task 1: Cleanup Levels at Other Sites

Rocky Flats Citizens Advisory Board
Radionuclide Soil Action Level Oversight Panel

April 1999

Submitted to the Rocky Flats Citizens Advisory Board
in Partial Fulfillment of Contract between RAC and RFCAB

"Setting the standard in environmental health"

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25

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26

CONTENTS

27

TASK 1: CLEANUP LEVELS AT OTHER SITES

INTRODUCTION

Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which action should be taken to prevent people from receiving unacceptable radiation doses. The soil action levels for radionuclides calculated for the Rocky Flats Environmental Technology Site (RFETS) by the U.S. Department of Energy (DOE), U.S. Environmental Protection Agency (EPA), and the Colorado Department of Public Health and Environment (CDPHE) have come under scrutiny because of lack of public involvement throughout their development. As a result of public concern, DOE provided funds to the Rocky Flats Citizen's Advisory Board to establish the Radionuclide Soil Action Level Oversight Panel (RSALOP) and to hire a contractor to conduct an independent assessment and calculate soil action levels for Rocky Flats. *Risk Assessment Corporation (RAC)* was hired to perform the study.

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

SOIL ACTION LEVELS AND CONCENTRATIONS AT OTHER SITES

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

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

The sites evaluated in this analysis are

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

Rocky Flats Environmental Technology Site

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

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

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

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

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

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

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

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

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

29

Hanford, Washington

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

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

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

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

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

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

Nevada Test Site

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

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

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

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

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

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

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

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

U.S. Nuclear Regulatory Commission DandD Code Scenarios

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

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

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

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

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

Johnston Atoll, Marshall Islands

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

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

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

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

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

Enewetak Atoll, Marshall Islands

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

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

Land use		
Food gathering	Agricultural	Residential
160	80	40

Maralinga, Australia

At the Maralinga Range in Australia, soil concentrations were calculated for a population of semi-traditional aboriginal people permanently residing in the area. Soil concentrations were calculated for a publicly accepted dose limit of 500 mrem. The only pathway considered in this analysis was exposure via inhalation of contaminated soil. The scenario from the Rocky Flats analysis most comparable to the Maralinga soil concentrations is the hypothetical future resident.

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

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

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

Semipalatinsk Nuclear Range, Kazakhstan

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

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

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

Thule, Greenland

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

Palomares, Spain

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

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

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

33

Summary of Available Site Information

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

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

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

SENSITIVITY ANALYSIS

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

METHOD OF COMPARISON

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

Each ratio is identified in two ways:

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Hanford, Washington

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

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

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

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

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

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

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

37

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

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

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

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

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

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

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

Nevada Test Site

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

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

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

39

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

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

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

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

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

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

U.S. Nuclear Regulatory Commission DandD Code Scenarios

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

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

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

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

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

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

Johnston Atoll, Marshall Islands

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

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

41

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

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

where

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

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

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

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

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

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

The first step was to determine the difference between dose conversion factors for the two sites. To extract the Johnston Atoll dose conversion factor from the existing information, we used an equation for effective dose from inhaled material. Equation (2) calculates dose (in units of millirem) from inhaled material.

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

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

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

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

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

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

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

43

Enewetak Atoll, Marshall Islands

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

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

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

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

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

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

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

The Bair Committee approved of the ERDA Operating Plan cleanup criteria and suggested that more specific guidance be established for the soil concentrations between 40 and 400 pCi g^{-1}. When the 1977 EPA guidance on transuranics was released, the Bair Committee adapted its recommendations for agricultural land soil concentrations to 80 pCi g^{-1} and food gathering land soil concentrations to 160 pCi g^{-1}. These values were apparently based on a dose assessment study performed by Lawrence Livermore National Laboratory. A first study done by Lawrence Livermore National Laboratory was based on the original soil cleanup criteria, but the results were deemed incorrect because of a mathematical error. The Laboratory performed a new dose assessment. Results from this new dose assessment influenced the Bair Committee's decisions concerning action levels for different soil uses.

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

Maralinga, Australia

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

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

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

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

where

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

and

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

45

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

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

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

where all quantities are as previously defined.

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

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

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

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

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

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

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

Semipalatinsk Nuclear Range, Kazakhstan

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

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

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

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

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

Thule, Greenland

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

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

Palomares, Spain

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

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

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

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

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

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

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

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

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

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

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

CONCLUSIONS

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

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

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

49

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

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

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

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

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

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

51

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53

RSALOP

Radionuclide Soil Action Level Oversight Panel

RSALOP COMMENTS

DRAFT REPORT: TASK 1: CLEANUP LEVELS AT OTHER SITES

LeRoy Moore

- I understand that you intend to put a table near the beginning of the paper in which you show the RSALs for different sites in terms of pCi of Pu/g of soil. I think to the right of these numbers it would help to have another column showing the dose to which the RSAL corresponds. You may also want a final column showing the targeted individual or residential scenarios. In some way or other, please put this on the table too, even though later in your test (at the top of Page 4), you say explicitly that a residential scenario is being used for all sites.
- On Page 3, where you introduce the two ways of referring to the ratio of RSAL to dose, there needs to be an elaboration or illustration – and it's probably best to elaborate with numbers from Rocky Flats; e.g., 1429 divided by 85 = 17.

Joe Goldfield - Since I have not had the time to make a comprehensive study of the report, the following comments are selective in nature:

- The original data from which calculations are made to develop various ratios comparing soil cleanup standards must be presented. The original data cannot be derived from the ratios. In each case, the original data has great significance because the soil action level and the mRem health effect on which it is based represents some group's best effort to present a scenario that is protective of the health of individuals living on that soil.
- I don't understand why data from the cleanup of Enewetak Atoll is not presented. The document entitled *"The Radiological Cleanup of Enewetak Atoll"* describes two proposed standards for cleanup of plutonium in soil – suitable for residential occupancy: one by Lawrence Livermore Laboratory of 10 pCi/g, and the other by the Bair Committee of 40 pCi/g.
- No mention is made of the soil cleanup standard set in 1976 by the Colorado Department Health of 1 pCi/g, nor of the standard suggested in Iggy Latour's paper, December 1995, of 3.8 pCi/g.
- A report by Sandia National Laboratory personnel (January 1998) for NRC proposed a standard of 2.15 pCi/g of plutonium in soil.
- No mention is made of the Working Draft Guide USNRC (August 1994) which proposed a soil cleanup standard of 1.89 pCi/g for an exposure of 15 mRem.

Submitted to *Risk Assessment Corporation* by Carla Sanda, Advanced Integrated Management Services, Inc. - March 11, 1999

DRAFT REPORT

Task 2: Computer Models

Rocky Flats Citizens Advisory Board
Rocky Flats Soil Action Level Oversight Panel

March 1999

Submitted to the Rocky Flats Citizens Advisory Board
in Partial Fulfillment of Contract between RAC and RFCAB

"Setting the standard in environmental health"

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56

DRAFT REPORT

Task 2: Computer Models

Rocky Flats Citizens Advisory Board
Rocky Flats Soil Action Level Oversight Panel

March 1999

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Submitted to the Rocky Flats Citizens Advisory Board
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57

CONTENTS

REVIEW OF THE RADIONUCLIDE SOIL ACTION LEVELS AT THE ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE

TASK 2. COMPUTER MODELS

Abstract

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

59

2. SOIL ACTION LEVELS

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

The soil action levels depend on four things:

(1) Predicted movement of the radionuclides through environmental media and into potential contact with people (environmental transport models and pathway analysis)
(2) Possible patterns of contact that hypothetical people are assumed to have with the radionuclides in the near or distant future; also, physiological characteristics that would affect the estimation of radiation dose that these hypothetical people would receive (exposure scenarios)
(3) Dosimetric models and data, including radionuclide-specific internal dose coefficients and dose rate factors for external exposure to gamma-emitting radionuclides; these models and data are used to estimate radiation dose to any hypothetical individual with known exposure to radionuclides in the environment (radiation dosimetry)
(4) Annual radiation doses that express protective thresholds for people who might be exposed to the radionuclides (annual dose limits).

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

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

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

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

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

DRAFT

61

2.1 Formulation

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

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

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

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

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

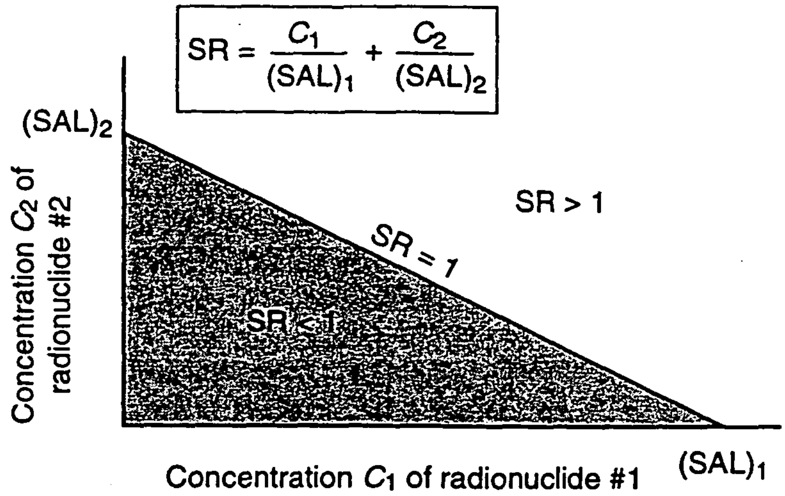

Figure 2.1-2. Geometric interpretation of the sum of ratios (SR) for two radionuclides (N = 2) and one scenario (S = 1).

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

$$D_{smi} = T_{smi} \cdot C_i . \qquad (2.1\text{-}2)$$

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

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

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

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

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

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

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

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

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

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

DRAFT

63

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

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

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

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

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

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

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

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

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

and the sum of ratios for scenario s as

DRAFT

64

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

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

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

2.2 Stochastic SALs

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

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

DRAFT

65

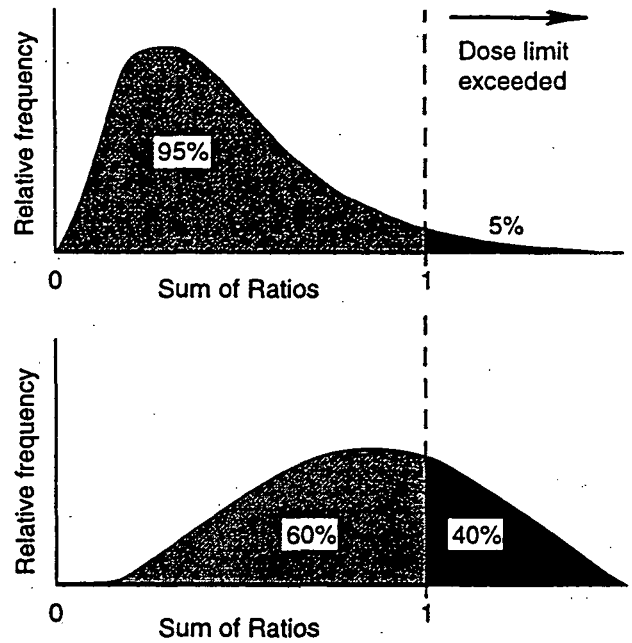

Figure 2.2-1. Schematic illustration of uncertainty distributions for the sum of ratios of soil concentrations divided by the corresponding soil action levels. In the top panel, the probability is 5% that the dose limit for a scenario would be exceeded. In the bottom, the probability is 40%.

2.3 Exposure scenarios

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

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

For the RFSAL assessment, some of the parameters are breathing rates for various activity levels and ages, soil ingestion rates for children and adults, fraction of time spent indoors and outdoors, and the potential use of or exposure to contaminated water from the area. Selecting appropriate parameters for the scenarios depends upon a thorough review of the scientific literature and fully considering the uncertainty (or variability) distributions of the relevant parameters. We use a wide range of references and studies to compile information on parameters. Subsequently, we can generate a distribution of values and sample from the distribution, using Monte Carlo techniques. This process considers the available studies equally. The distributions can be characterized with a central value such as the median and some measure of the spread of the distribution, such as the standard deviation or the 5th and 95th percentiles of the distribution. In developing a particular scenario and considering variability of a parameter within the population studied, we can use a high (or low) percentile of the distribution as needed to extend protection to a larger fraction of a potentially exposed population with characteristics similar to those of the scenario subject. Once a parameter value is selected from our distribution of values for use in the scenario, the scenarios are considered fixed just as standards are fixed as a benchmark against which to measure an uncertain value. Behavioral characteristics should be plausible and relevant to the exposure situations and the radiation protection objectives.

Scenarios provide a technical basis for focusing on those pathways and characteristics that are most important in the dose assessment. For example, for plutonium in soils at Rocky Flats, the inhalation pathway will likely prove important. The inhalation or breathing rate affects the transport of airborne contaminants to the respiratory tract and also influences their deposition onto surfaces of the airways and in the pulmonary region. As a result, it is important to exercise care in selecting breathing rate values for each scenario. We have compiled data from numerous published papers to provide perspective in the selection of suitable breathing rates. The selection of input parameters will be described fully in the Task 3 report for this project. The historic approach for estimating breathing rates over a specified time period is to calculate a time-weighted-average of ventilation rates associated with physical activities of varying time durations. A second approach for determining breathing rates for various populations is based on basal metabolism and measured food-energy intakes and energy expenditures. In general, breathing rate studies indicate that gender makes little difference on breathing rates through about age 12. For teens through adulthood, the breathing rate can be 40-50% higher in males than females. There is also age dependency on breathing rates, with adults having breathing rates that are about a factor of 3 higher than for young children. For a person of a given age and gender, the most significant parameter affecting breathing rate is the level of activity: breathing rates can be 15 times higher under maximum work conditions than resting. This activity dependence is important for acute exposure of a few hours, but less important for continuous chronic exposure (year). Because of the variability in breathing rates with activity level and age, it is more defensible to use a distribution of values from which to select the input breathing rates (using a high percentile, for example) for an individual scenario.

The use of reference scenarios also allows the consideration of special cases of interest. For example, we can see the impact of soil ingestion on dose by varying the exposure from contaminated soil particles on garden foods consumed by everyone to contaminated dirt ingested by a child.

DRAFT

67

RAC is evaluating the three scenarios described in the report, *Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement*, dated October 31, 1996 (DOE/EPA/CDPHE 1996), along with additional scenarios that we have proposed and described at the monthly Radionuclide Soil Action Level meetings. *RAC* believes strongly that it is important to describe the process behind the development of the scenarios, to provide the panel with a broad range of scenarios for evaluation, and to consider a number of likely scenarios before final scenarios are selected for the project. In our discussions with the panel, we have used several breathing rate studies as examples of the kinds of data that will be used to develop uncertainty distributions for key parameters. In these meetings, we described the step-wise process to show how breathing rates can be selected based on activity levels and age, and how these values are summed over a specified time period (e.g. hour, day or year) to yield an annual breathing rate. This demonstration was important to understand that an annual inhalation rate for an airborne radionuclide is based on a weighted average rate, where the weights are determined from the times spent in different activities and at indoor or outdoor locations throughout the day.

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

1. The future residential exposure scenario assumes that an individual resides onsite all year and grows and consumes homegrown produce. This person would be exposed to radioactive materials in soils by directly ingesting the soils, by inhaling resuspended soils, by external gamma exposure from contaminated soil and airborne radioactivity, and by ingesting produce grown in contaminated soil. This scenario is from the current Rocky Flats Cleanup Agreement.

2. The open space exposure scenario assumes the person visits the site 25 times per year for recreational purposes, spending 5 hours per visit at the site. The person would be exposed to radioactive materials in the soil by directly ingesting the soils, by inhalation of resuspended soils, and by external gamma exposure from the soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement.

3. The office worker exposure scenario represents an individual who works a 40-hour per week, 50-week per year job indoors in a building complex at the site. It is assumed that this person would be exposed to radioactive material in soils by directly ingesting the soils, by inhaling resuspended soils, and by external gamma exposure from soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement.

4. The resident rancher scenario assumes future loss of institutional control. The rancher is raising a family, maintaining a garden and leading an active life at the site, spending 23 hours per day, 365 days per year or 8400 hours at the site. Of that time, over 40% is spent out of doors. The potential pathways of exposure for this person include inhalation; eating produce

68

from garden irrigated with some water from a stream on the site, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils and airborne radioactivity. The annual breathing rate is 10,000 m^3 per year, based on a time-weighted average of breathing rates and activity levels as described during the monthly RSALs meetings. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.

5. Infant in rancher family is 0 to 2 years of age, and onsite 23.5 hours per day, 365 days per year, or 8600 hr/year. The infant's potential pathways of exposure include inhalation, some ingestion of produce from family garden, some direct soil ingestion from outdoor activities, and direct gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
6. The child of the rancher family is assumed to be 5 to 17 years of age, and onsite 16 hours per day, 365 days per year, or 5800 hr/year. The potential pathways of exposure include inhalation, eating produce from garden irrigated with water from a stream on the site, direct soil ingestion, and gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
7. The office worker scenario is quite similar to the office worker scenario already described in the current Rocky Flats Cleanup Agreement. The differences are a higher breathing rate of 200 m^3 per year and a higher soil ingestion rate of 25 g $year^{-1}$. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
8. The recreational land user is similar to the open space user already described in the current Rocky Flats Cleanup Agreement. The differences are more frequent site visits (100 times per year for 3 hours per visit), a higher annual breathing rate of 750 m^3 per year, and a higher soil ingestion rate of 25 g $year^{-1}$. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
9. The subdivision resident lives in a developed neighborhood, works in a home office on the site, maintains a garden for fresh produce, and uses the site for running or biking for physical exercise. The person is onsite 22.5 hours per day, 350 days per year, or 7900 hours per year. Of that time, the person is outdoors 15% of the time. The annual breathing rate (7400 m^3 per year) and soil ingestion rate (88 g $year^{-1}$) are slightly higher than the residential scenario described in the current Rocky Flats Cleanup Agreement. *RAC* proposed this scenario for consideration at the February 1999 RSAL meeting.
10. The current onsite industrial worker scenario assumes a person works onsite 8½ hours per day, 5 days per week, 50 weeks a year, or 2100 hours per year. It is assumed that 60% of the worker's time is spent outdoors. The potential pathways of exposure for this person include inhalation, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils. The annual breathing rate is 3600 m^3 per year, based on a time-weighted average of breathing rates and activity levels for the time spent onsite. *RAC* proposed this scenario for consideration at the February 1999 RSAL meeting.

DRAFT

69

DRAFT

Table 2.3-1. Key Parameter Summary for the Rocky Flats Environmental Technology Site Radionuclide Soil Action Levels

Parameter	Current DOE/EPA/CDPHE Scenarios: Residential	Open space	Office worker	Additional Scenarios for consideration: Resident rancher	Infant of resident rancher (NB-2 yr)	Child of resident rancher (5-17 yr)	Office worker	Rec. land user	Neighborhood resident	Current site industrial worker
Time on the site (hr/day)				23	23.5	16	8	3	22.5	8.5
Time on the site (hr/yr)	8400	125	2000	8400	8600	5800	2000	300	7900	2100
Time indoors onsite (hr/yr)				4700	7740	5075	1750	300	6700	900
Time indoors onsite (%)	100	100	100	57	90	88	88	100	85	40
Time outdoors onsite (hr/yr)	0	0	0	3700	860	725	250	0	1200	1200
Time outdoors onsite (%)	0	0	0	43	10	12	12	0	15	60
Inhalation Shielding Factor	1	1	1	1	1	1	1	1	1	1
Breathing rate (m^3 per year)	7000	175	1660	10000	1800	4400	2000	750	7400	3600
Soil ingestion (grams per day)	0.2 for 350 d	0.1/ visit for 25 visits/yr	0.05 for 250 days	0.25 for 365 days	0.04 for 265 days	1 for 365 days	0.1 for 250 days	0.25 / visit for 100 visits	0.25 for 350 days	0.25 for 250 days
Soil ingestion (grams per yr)	70	2.5	12.5	90	15	365	25	25	88	62
Irrigation water source	na	na	na	Woman Creek	Woman Creek	Woman Creek	na	na	groundwater	na
Irrigation Rate (meter/yr)	na	na	na	1	1	1	na	na	1	na
Fraction of contaminated homegrown produce	1	0	0	1	0	1	0	0	1	0

na = not applicable

3. SITE CONCEPTUAL MODEL

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

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

It is important to be as specific as possible about the nature of the models that simulate the movement of the radionuclides along the environmental pathways leading to possible exposure of people. There is no unique approach to the definition of these models: they can range from simple to complicated. The choice of definitions is usually indicated by experience, consideration of the site, and what is mathematically or computationally tractable. Pathways that can be shown to contribute negligibly to the endpoint of the calculation, relative to other pathways, can be omitted, but this must be done with care. Section 3.1 describes the pathways that are potentially relevant to the RFETS. The pathways depend on the exposure scenarios, which we described in Section 2.3. The models, coupled into a system, are treated as uncertain (principally through their parameters: *parametric uncertainty*), and when we are given a set of measured or hypothesized concentrations of radionuclides in the soil, we apply Monte Carlo analysis to the sum of ratios to derive a distribution that tells us the probability that the dose limitations will be met.

3.1 Transport pathways

3.1.1 Availability of residual radioactivity in surface soil over time

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

Our approach to multimedia modeling emphasizes the effort to preserve mass balance and to avoid deliberate biasing of environmental concentration estimates. This approach goes hand in hand with our treatment of uncertainty distributions. An example of an approach that would violate this principle is to estimate loss of radioactivity from surface soil by runoff and leaching without accounting for the complementary depletion of radioactivity in the surface soil reservoir. Such calculations can be defended as conservative, but the loss of mass balance accounting generally introduces difficulty into the analysis and interpretation of uncertainty, and we prefer to avoid this difficulty. Our alternative is to try to put the conservatism into the uncertainty distributions, preserving mass balance and minimizing bias. We stress that these are general guidelines; sometimes they cannot be adhered to as closely as one would like.

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

DRAFT

72

is sometimes questionable in the interpretation of process modeling. Narrower definitions of K_d are used in laboratory work, and criticisms of environmental soil modeling often turn on the use of this parameter and its different interpretations (Jirka et al. 1983).

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

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

3.1.2 Spatial disaggregation of soil

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

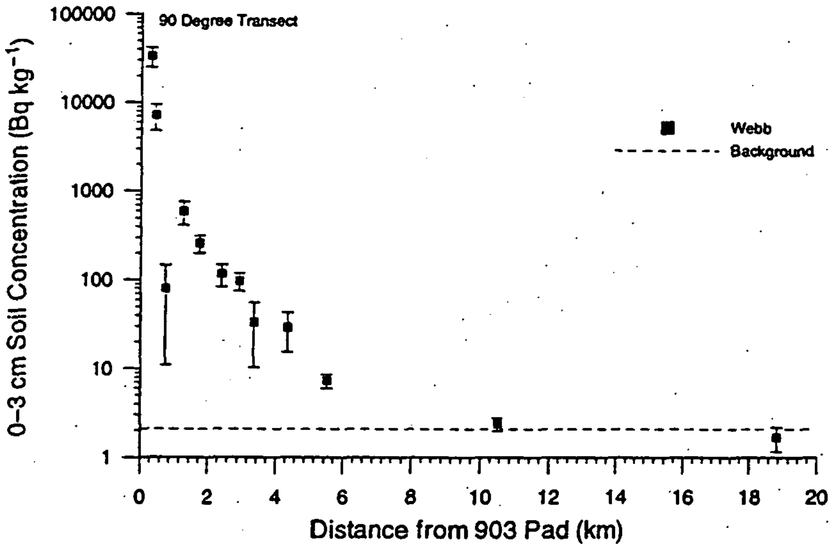

Figure 3.1.2-1. Plutonium-239 concentrations in soil (Bq kg^{-1}) at RFETS along a 90° transect (eastward) from the 903 pad area. The data are from Webb (1996).

3.1.3 Resuspended contaminated soil

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

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

DRAFT

74

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

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

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

All three of these approaches require quantification from the analyst or from default values or formulas supplied by the programs. In this respect, the mass loading approach is perhaps the most direct, requiring as its parameter the very air dust concentration that we seek to estimate. The parameter estimate should be based on measurements taken at the site and averaged over as long a period as possible. The measurements of Langer (1986) indicate a mean total dust concentration of 47 μg m^{-3} with standard deviation 9.0 μm at the 1-m height for the period November 1982 through December 1984. This total quantity, however, includes a substantial

75

fraction of particulate mass in a size range that is not regarded as respirable (59%). If the coarsest category of particles is discarded, the mean concentration is only 19.2 $\mu g\ m^{-3}$. Most of the resuspended plutonium activity (81%) at the 1-m level is associated with the coarse (non-respirable) particles, leaving only 19% associated with respirable particles. We cite these data to illustrate the point that one should consider the question of the size distribution of the airborne dust and the distribution of plutonium activity over the airborne particles in order to make credible estimates of inhalation dose. The computer programs that implement mass loading do not exercise this judgment, although default values of some parameters may be supplied. Another complication is that air samplers lose efficiency as the particle aerodynamic diameter increases, and the efficiency loss is aggravated by the high wind events that cause much of the resuspension. Thus the measurements taken at Rocky Flats are subject to uncertainties of interpretation, and these uncertainties need to be quantified and incorporated into the calculation.

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

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

3.1.4 Groundwater and surface water transport

In calculating the proposed soil action levels (DOE/EPA/CDPHE 1996), the groundwater and surface water pathways were dismissed because (1) surface water features (Woman and

DRAFT

76

Walnut Creeks) on the site are perennial and would not provide a reliable year-round water source for an individual living on the site and (2) surface aquifers underlying the site do not produce enough water for domestic or agricultural use. In addition, the aquatic food pathway was eliminated because the streams are not capable of sustaining a viable fish population. In this section, we will discuss these assumptions and the rationale behind them, and we will examine the ramifications of dismissing the groundwater and surface water pathways in the assessment.

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

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

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

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

The LAHU is composed of fine to medium grained sandstone separated by a few claystone beds in the upper portion. Thickness ranges from 200 to 220 feet for the "A" and "B" sandstone that comprise the lower interval of the Laramie formation, and 80 feet for underlying Fox Hills sandstone unit. The Laramie–Fox Hills aquifer system is the target of most water wells in the vicinity of Rocky Flats because this aquifer provides sufficient water for domestic and industrial uses. Recharge to the aquifer takes place along the foothills west of the RFETS where the permeable sandstone beds of the formation are folded up and exposed. The permeable sandstone generally dips eastward toward the center of the Denver Basin.

Surface water features at the RFETS include Walnut and Woman Creeks and several ditches that provide irrigation water. Walnut and Woman Creeks are perennial and generally respond to seasonal fluctuations in precipitation, recharge, groundwater storage, and stream and ditch flow. These creeks drain into Great Western Reservoir and Standley Lake.

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

It is well beyond the scope of this project to address the groundwater pathway in any substantial way other than through a simple screening exercise. Sophisticated groundwater modeling is difficult and time consuming, requiring substantial quantities of field data to characterize subsurface hydrologic units. We examine a conservative calculation in order to address the question of whether or not the pathway can be ruled out of the current analysis. We activate the groundwater pathway model in the RESRAD simulations, using the site conceptual model and parameter values developed and documented in the proposed soil action level document (DOE/EPA/CDPHE 1996). The RESRAD conceptual site model assumes that a receptor uses groundwater derived from the UHSU for drinking water only. No irrigation or livestock watering was assumed. The default RESRAD water ingestion rate of 510 liters per year was used in the analysis. Parameter values used in the assessment were reviewed and appear to be reasonable based on the information provided in the hydrogeologic characterization reports (DOE 1995).

Results for Tier 1 Action Level (85 mrem) residential exposure scenario are shown in Table 3.1.5-1. Note that action levels changed only for ^{241}Am, ^{241}Pu, and ^{234}U. In these cases transport of a mobile radioactive decay product with a relatively high ingestion dose conversion factor is what caused groundwater ingestion doses to outweigh doses from external sources and inhalation.

For the radionuclide given the most attention (^{239}Pu), the soil action level remained unchanged. The results of this exercise suggest that the rationale for dismissing groundwater as a viable pathway should perhaps be investigated further. The ongoing activities of the Actinide Migration Panel and other studies involving plutonium mobility should shed additional light on

18 **DRAFT**

this subject. However, for the purpose of calculating soil action levels, we will ignore the groundwater pathway, keeping in mind that all results are subject to reinterpretation based on any new findings from actinide migration studies and additional investigations performed for site remediation purposes.

Table 3.1.5-1. Soil Action Levels for the Residential Exposure Scenario at the 85 mrem Level Including and Excluding the Groundwater Pathway

Radionuclide	Soil action level without groundwater pathway ($pCi\ g^{-1}$)[a]	Soil action level with groundwater pathway ($pCi\ g^{-1}$)
241Am	215	110
238Pu	1529	unchanged
239Pu	1429	unchanged
240Pu	1432	unchanged
241Pu	19830	3370
242Pu	1506	unchanged
234U	1738	660
235U	135	unchanged
238U	586	unchanged

[a] Source: DOE 1996.

3.2 Exposure Modes

The exposure modes described in this section have already been mentioned in previous sections to illustrate exposure pathways. The basic modes are inhalation and ingestion (internal exposure) and exposure to an external medium containing beta- and (primarily) gamma-emitting radionuclides. Other possible modes for internal exposure are absorption of a radioactive compound through intact skin or introduction of radioactivity into blood through injection or by contact of a radioactive chemical with an open injury.

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

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

Dose can be estimated for any organ that absorbs energy from ionizing radiation. The *effective dose* is a concept promoted by the International Commission on Radiological Protection (ICRP), which gives a nonlocalized definition of dose that is roughly proportional to the risk of radiation-induced cancer in *some* organ or tissue; the proportionality is achieved by weighting the equivalent dose to each internal organ with a relative risk coefficient for the organ (ICRP 1977). The effective dose is not to be confused with whole-body dose, which lacks this more refined connection to cancer risk.

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

DRAFT

80

4. CANDIDATE COMPUTER PROGRAMS

4.1 Introduction

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

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

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

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

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

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

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

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

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

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

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

4.1.1 Verification of Computer Programs

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

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

4.1.2 Validation of Computer Programs

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

82

mathematical models encoded in the computer program to predict the behavior of contaminants in the environment.

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

The interpretation of validation exercises is never entirely clean. Consider once again the example of predicting uranium concentrations in air. Our calculations involve more than the computer program: there are the estimates of the uranium releases, which are subject to error, and there are meteorological data, which may or may not be accurate for the locations and period for which they were applied. It is possible for errors in the data to compensate for errors in the model, giving apparently good results and encouraging us to trust a program that intrinsically might not be an acceptable representation of the processes we are simulating. Alternatively, errors in the data could make an acceptable model look bad. When we must depend on data that are available, it is practically impossible to implement rigorous designs that might remove these confounding effects. We must generally be satisfied with making as many tests of two or more correlated functionalities (e.g., diffusion and deposition, if we have data for both) as possible, in the hope that good agreement of predictions and data will be persuasive at an admittedly subjective level.

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

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

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

Given the inherent difficulties of validation, one often has to supplement it with other approaches. Uncertainty analysis, appropriately applied, leads to results that quantify possible errors that derive from lack of knowledge or variability of parameters. Uncertainties about the proper structure of the model are more difficult. The temptation is to try to broaden the "space" of models from which the one in question has been drawn and to extend the uncertainty calculation to a representative set of possible replacements from this space of models (Draper 1995). But this approach has immense conceptual and technical difficulties. A more pragmatic option is to accept model structures that have been affirmatively validated in a variety of similar problems as provisionally correct but with magnitudes of uncertainty indicated by a broad range of experience. For example, in atmospheric diffusion calculations, the straight-line Gaussian plume model is widely used in environmental applications, although this model is based on assumptions that are technically too simple for most of those applications. But experience and experiment indicate that for particular categories of predictive use, the Gaussian plume can be associated with corresponding uncertainty distributions. For example, from a review of numerous sets of experimental data, Miller and Hively (1987) concluded that for flat terrain, away from coastal areas, the Gaussian plume can predict annual averages of concentrations within a factor of two 90% of the time out to a distance of 10 km and within a factor of four with 90% probability somewhat beyond that distance. Such information must be applied with care and skill, but it provides an empirical representation of atmospheric diffusion and some level of confidence in the model; the cost is the stated uncertainty. This illustration, however, should not be interpreted to mean that the straight-line Gaussian plume model is applicable with knowable uncertainty to any atmospheric diffusion problem. It is not, and we know of no model that is.

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

4.2 RESRAD

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

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

4.2.1 RESRAD overview

The manual for Version 5.0 (Yu et al. 1993), which was distributed with Version 5.82, does not correspond to the more recent graphic user interface (GUI) implementation. A user's guide for the latter, which is a replacement for Chapter 4 in the manual (Yu et al. 1993) is now available from ANL or from the web site http://www.ead.anl.gov/resrad. DOE has directed ANL to discontinue distribution of RESRAD versions for the DOS operating system, the most recent of which was Version 5.62. Some of the information we received seemed to suggest that there might be incompatibilities of DOS versions with contemporary Windows operating systems. However, we have tested Version 5.61 in a command window under Windows NT and encountered no problems with it. However, a major algorithmic change affecting the Windows versions of RESRAD (beginning with Version 5.75) has been made in the area factor for the resuspension of soil particles (Chang et al. 1998). The difference in predicted doses and soil action levels can be significant. We will discuss the change in a later section.

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

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

Radioactivity from the contaminated zone may be resuspended by a mass-loading model; separate resuspension pathways are implemented for inhalation exposure and for foliar deposition on crops and animal fodder. External doses from exposure to gamma emissions from the contaminated zone and the resuspended contaminated soil particles are estimated. Beginning with Version 5.60, the external radiation field calculations incorporated refinements for the finite area and volume (with possibly irregular shape) of the contaminated zone, in contrast to previous methods that assumed semi-infinite distributions of radioactivity in source media (Kamboj et al. 1998).

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

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

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

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

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

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

4.2.2 Code acquisition

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

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

If the questions of algorithmic inconsistency between the RESRAD documentation and the program can be resolved satisfactorily, we believe RESRAD can be used as the primary tool for investigating the benchmark (and possibly other) scenarios of use of the Rocky Flats site and the establishment of the relationship between radionuclide levels in the soil and annual dose

DRAFT

98

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

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

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

4.2.3. Changes in the area factor for resuspension

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

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

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

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

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

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

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

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

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

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

90 **DRAFT**

discrepancies, and higher wind speeds would give larger ones. Larger areas would correspond to better agreement between the two area factors. Figure 4.2.3-1 shows a comparison of the old and new area factors for particle diameter 1 µm plotted against $\sqrt{A}$ for several values of the wind speed.

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

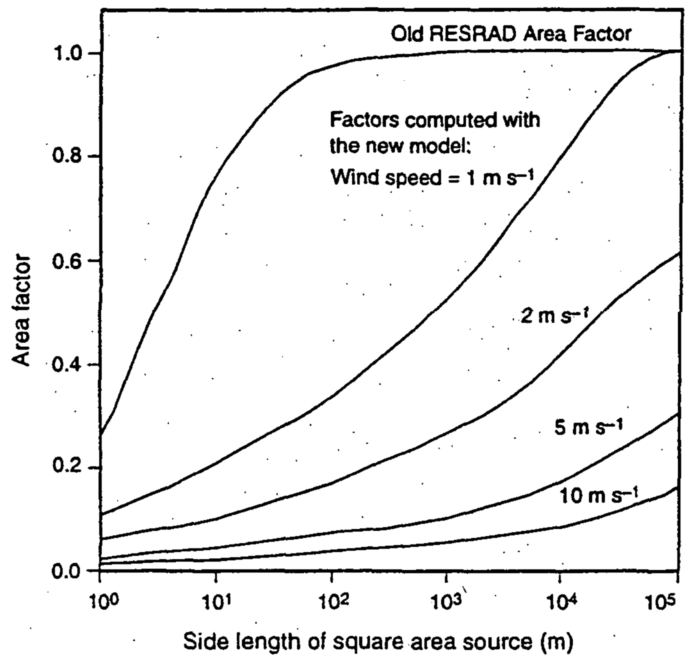

Figure 4.2.3-1. Comparison of the old and new RESRAD area factors for particle size 1 µm, plotted against the side length of a square contaminated area. The new area factor is shown for several values of the wind speed. This figure was redrawn from Chang et al. (1998).

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

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

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

It is not our intent to criticize the RESRAD developers. The models and parameters that they have applied to estimate the area factor are well known and frequently invoked. Their approach is rational from a research standpoint, their analysis seems thorough, and we are appreciative of the well-organized numerical explorations they have provided in Chang et al. (1998). Our reservations have more to do with objections to generic application of assessment models. The developers consider this formulation of the area factor more realistic than the older version that was based on a simple box model (Equation 4.2.3-1), and in a sense that may be true. But in any assessment, the analyst should be weighing the appropriateness of any factor that enters into the calculations for the site in question and integrating each factor into the composite uncertainty picture. As a practical matter, we find it more difficult to circumvent this formulation of the area factor when we execute RESRAD Version 5.82. We certainly agree with the last sentence in Chang et al. (1998): "However, if measurement data are available, the measured air concentrations [*sic*] data should be used in RESRAD analysis." The user's manual should clarify just how this is to be done; we assume it would involve supplementary off-line calculations based on RESRAD output. We will be making use of such measurements in the calculations for Task 5.

In general, one can expect Versions 5.75 and newer of RESRAD to predict lower annual resuspension-dependent doses and correspondingly larger radionuclide soil action levels, with the extent of the discrepancy depending on the values supplied for the mean wind speed and the area of the contaminated zone. For application to the Rocky Flats site, we cannot make a more definite statement at this time, until an appropriate area for the field of contamination is determined. In regard to the version of RESRAD that will be applied, there is some ambiguity about the intentions of the regulatory agencies. The soil action level document

92 **DRAFT**

(DOE/EPA/CDPHE 1996) presents RESRAD parameters and computed soil action levels that appear to correspond to an earlier version of the code (perhaps 5.61 or 5.62). This was probably the most recent version available at the time that document was prepared. But if the assessment were to be carried out in a purely formal manner, with the newer version of the code being substituted and executed with the same set of parameters, the foregoing analysis indicates that a possibly important change in the predictions would occur.

4.3 MEPAS

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

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

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

4.4 GENII

At the direction of the U.S. Department of Energy in 1988, the Hanford Environmental Dosimetry Upgrade Project was undertaken by Pacific Northwest Laboratory to incorporate the internal dosimetry models recommended by the International Commission on Radiological Protection into updated versions of the environmental pathways models used at Hanford. The resulting second generation environmental dosimetry computer codes were compiled in the Hanford Environmental Dosimetry System — Generation II or GENII (Napier et al., 1988). The

GENII system was developed by means of tasks designed to provide a state-of-the-art, technically peer-reviewed, documented set of programs for calculating radiation doses from radionuclides released to the environment.

4.4.1 Code overview

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

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

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

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

The GENII package of codes was developed under a stringent QA plan based on the American National Standards Institute (ANSI) standard NQA-1 (ASME 1986) as implemented in the PNL Quality Assurance Manual PNL-MA-70[1]. All steps of the code development have been

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

documented and tested. Extensive hand calculations have been performed and are available for review on request

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

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

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

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

GENII also considers an on-site groundwater pathway like RESRAD. However, RESRAD computes transport from the source, through the vadose (unsaturated) zone, and into the aquifer while GENII only allows the user to input a previously measured or modeled groundwater concentration, and dose calculations are performed on that basis. In RESRAD, the groundwater model consists of relatively simple representations of subsurface aqueous flow and transport and does not consider off-site transport of contamination in the aquifer.

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

4.4.3 Code acquisition and testing

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

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

DRAFT

96

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

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

a. from DOE (1996), Attachment I

b. Area of contamination in GENII is only defined in terms of less than or greater than 1250 m^2

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

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

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

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

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

a. [(DSR (RESRAD) – DSR (GENII)]/DSR (RESRAD)

4.5 MMSOILS

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

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

DRAFT

98

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

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

4.6 DandD

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

4.6.1. Code overview

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

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

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

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

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

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

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

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

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

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

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

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

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

100 **DRAFT**

4.6.3. Code acquisition and testing

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

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

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

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

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

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

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

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

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

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

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

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

DRAFT

102

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

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

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

5. CONCLUSIONS AND RECOMMENDATIONS

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

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

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

When RESRAD is applied to the resuspension pathway, we recommend that it be with full awareness of the effect of the area factor. As we mentioned in Section 3.1.3, measured air concentrations of some of the radionuclides in the source term are available, and careful consideration should be given to using these measurements or calibrating the model to them. This approach may require manipulating the input parameters so that the area factor is effectively 1. Similar manipulations will be required if alternative resuspension models, such as the EPA models of Cowherd et al. (1985), are to be applied.

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

We have some recommendations for DOE and the developers of RESRAD. We are aware that the evolving Windows graphic user interface (GUI) is intended to make the program more accessible to a variety of users, but this greater utility comes at a cost to some potential users. It often is desirable to link programs together, with outputs from one becoming inputs to another. The procedure is usually implemented by writing scripts, which are control programs for the process (Unix operating systems are particularly hospitable to this approach). But a GUI defeats script-driven executions. We are not suggesting that the GUI be eliminated, but we do urge DOE and the RESRAD developers to facilitate a way of bypassing the GUI and launching RESRAD from the command line.

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

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

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

DRAFT

105

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106

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DRAFT

107

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June 23, 1999

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Dear Ms. Sanda:

Enclosed is a revised version of our responses to the reviewers comments for Task 2. This revision takes into account our discussion of June 7 and the helpful suggestions of LeRoy Moore and some of our own.

Sincerely,

John E. Till

John E. Till, Ph.D.
President

enclosure

109

RAC RESPONSES TO PEER REVIEWER COMMENTS

Task 2: Computer Models

Radionuclide Soil Action Level Oversight Panel

May 1999

"Setting the standard in environmental health"

RAC

Risk Assessment Corporation
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110

Responses to Reviewers' Comments on RSAL Task 2 Report

Since the reviewers are not openly identified by name, there is no satisfactory way to indicate which reviewer's comments we are responding to at any particular time. This situation thwarts a topical organization of these responses. Instead, we present the responses in five sections (one per reviewer), and we identify each reviewer by the number of pages in his or her printed copy (fortunately, no two reviewers produced copies of equal page length). In each reviewer's section, we respond to selected comments in the order in which they appear in the copy. References are placed at the end of the section in which they were called out.

Reviewer Two

This is a useful and helpful review. The reviewer is familiar with the Rocky Flats site and the history of radionuclides in the soil there. We will give serious consideration to all of this reviewer's suggestions.

2. It is extremely important to use every opportunity to apply site-specific data for soil concentrations and parameter values and their uncertainty distributions to the models that are chosen for the analysis. It is equally important to understand the inherent structure and workings of the models and to be able to modify them as necessary to make them relevant to Rocky Flats. The models should be both verified and validated to the extent possible.

3. I do not feel that *RAC* should limit their analysis to one or two models such as RESRAD or GENII. Other models that may have been used to develop soil action levels at Rocky Flats or elsewhere should also be examined in an effort to understand why such different numerical action levels have arisen. One recent report ("Recommended screening limits for contaminated surface soil and review of factors relevant to site-specific studies", NCRP Report 129, issued January 29, 1999) should definitely be consulted, for example. As a general philosophical point, the skill, knowledge and effort of the model user is often more important that the model itself in arriving at credible predictions.

These comments support *RAC*'s contention that this project should place less emphasis on specific computer programs and more on appropriate models (remembering that we are careful to distinguish between models and computer programs), data, and the knowledge and skills of the analyst. NCRP Report No. 129 was not available before about April 1 (at which time the work for this task was in its late stages). We are familiar with the report and are examining it for its relevance to this work.

5. The amount of resuspension of contaminants from the soil surface is dependent on many processes, both natural and anthropogenic. It is my experience that management of the land is a first-order determinant of resuspension, and this should be recognized and built into the various land use scenarios. Any form of human disturbance, especially anything which disturbs the natural vegetation cover, is bound to increase resuspension during high winds, as well as surface

111

> runoff following rainstorm events. Unpreventable phenomena that could cause major disturbances are fires, tornadoes, and floods. These should perhaps be considered by the *RAC* as stochastic events with a certain probability of occurrence. If any of these phenomena were to occur, then short to medium-term increases in resuspension or runoff, perhaps of dramatic proportions, could result.

This perceptive comment sets a potentially difficult task for this project. We expect to be able to check model predictions of resuspension against (at least) Langer's measurements in the 1980s, which provide two years of data, but which consider only the ground cover that existed at that time. A fire that denuded the landscape would increase resuspension by an unknown amount. A tornado that touched down near the site of the 903 pad would immediately send substantial quantities of contaminated soil and litter airborne, and the resulting disturbance of ground cover and surface soil would permit an enhanced resuspension of radioactivity until the previous state was restored. Credibly quantifying the aftermath of these events is very difficult. They can be discussed in the reports, but systematically incorporating them into scenarios would require a great deal more effort and debate than the stringent schedule of this project permits.

Reviewer Three

This reviewer appears to have missed some things in his or her reading of the report. Hopefully the responses below help to clarify these.

> . . . The review of the models, in general, seems sufficient with a few exceptions. The report lacks a clear, concise statement of the criteria used to identify the models that would be selected for review. This should appear in the Introduction.

Such a list of five criteria appears at the beginning of Section 4.1 (page 29). It could be replicated in the introduction, but the existing placement seems more appropriate to us.

> . . . In addition, *RAC* did not explicitly address the models' capabilities to address offsite exposures. This was explicitly mentioned in the RFP and *RAC*'s proposal of work and should be explicitly addressed in the review.

In the overview of GENII, Section 4.4.1, third paragraph, we find the following: "The proposed soil action levels developed for the RFETS are essentially based on a near-field scenario. The RESRAD code is not capable of addressing directly what GENII defines as a far-field scenario, and therefore, GENII appears to have an advantage as a model that may provide dose estimates to off-site individuals." Perhaps the point also deserves mention in the introduction to Section 4. With regard to offsite exposures, it will be pointed out that the approach we are taking to derive RSALs requires that people living farther away (i.e. offsite) will receive less exposure than those individuals who live on the area where the RSAL is being calculated. Therefore, "offsite" exposures are being taken into account implicitly.

> 1) Include a list of definitions of acronyms and variable names used in the equations.

112

We will consider this recommendation. If the reviewer means variables used in the equations, this could be done, but variable names in the programs run into the hundreds and including them would be difficult.

2) The second paragraph of the introduction requires clarification. In order to "...make clear our [*RAC*'s] conception of the task to which the programs would be applied...", *RAC* provides a vague definition of SALs. The introduction should be where a succinct, readily understandable definition is provided. I suggest:

We will reexamine the definition and decide whether we believe it requires further work. As part of an effort to make the technical reports more understandable, we intend to include a layman language summary at the beginning of each report. Hopefully this will help the non-technical reader understand the project better.

3) In the detailed discussion of the use [of] SR (Section 2), it should be emphasized that the use of the SR is predicated on the assumption that the model estimated radiation dose is linear to the initial radionuclide concentration in soil. It is important to ensure that this is true for the models reviewed.

This condition is set forth as Equation 2.1-2. Few assessment models are implemented with nonlinear dependence of committed dose (the end point of these predictions) on environmental concentrations. If the reviewer knows otherwise, we would appreciate knowing more about them.

4) In eq. 2.1-1, it seems to me that there is no reason to include scenario as an index. It confuses the discussion. In addition, EPA and et al. have traditionally kept exposure scenario- and dose limit -specific SALs separate (e.g., Table 5-1 in US DOE, 1996). When a particular SAL is selected for a site, it seems sufficient to indicate that the selected SAL is or is not protective of whatever other exposure scenario/dose limit combinations have been evaluated.

In our analysis, a scenario corresponds to a single individual. Thus the rancher, his wife, and his child would ideally be implemented as three correlated scenarios. However, we acknowledged that "as a practical matter, we may wish to treat different scenarios as if they were independent" (page 9, parenthetical remark in the next-to-last sentence).

5) I am not sure how the soil action levels "represented as a joint probability distribution" that *RAC* proposes developing should be interpreted in field applications. After all, the purpose of SALs is to be useful in the field, i.e., to provide either a means of determining the acceptability of measured radionuclide concentrations and/or a quantifiable remediation goal. How will measured concentrations be compared to SALs specified as joint distributions (i.e., compare means, variances, and correlation coefficients?—what if mean is the same, but variance or correlations are different?) I think SALs are more appropriately expressed deterministically for comparison to mean measured contaminant concentration levels, as described in Yu et al.(1993) for sites with homogeneous contamination (1993,see p.33-34, and especially see eq. 3.4. Note

> that there is a separate discussion on how to handle inhomogeneous contamination on p. 35). (In addition, RESRAD (Yu et al., 1993) (and likely the other models ??) assume uniform initial contaminant concentrations in the contaminated soil layer. This is, to be sure, a simplification of reality. When contaminant concentrations are not uniform, the deterministic initial contaminant concentrations input to the model can most appropriately be interpreted as the spatially-weighted mean contaminant concentration. There is, to be sure, some uncertainty in this mean due to sampling statistics. However, this uncertainty can be minimized by an adequate sampling strategy. I would caution against thinking that applying an uncertainty distribution to the input initial contaminant concentration would account for variability of contaminant concentrations in the contaminated layer.)

We explained our recommendation for using the distribution of the sum of ratios as an action level criterion (Section 2.2 and Fig. 2.2-1). There is nothing in the formulation to preclude handling the concentrations as constants, if everyone is satisfied that this approach is justified by estimates of sampling error and consideration of possible uncertainties in the representations of the concentrations as spatial averages. We deliberately left this choice open. However, the SALs in the denominators of the ratios are still uncertain, and the sum of ratios needs to be treated as a distribution. It appears that the reviewer may uncomfortable with the idea of applying uncertainty to environmental assessments. Perhaps the reviewer is just expressing caution with regard to including uncertainties in the analysis, and this is a valid point. There is no question that the document of Yu et al. describes deterministic models, and RESRAD was designed to implement such models. Nevertheless, we do not believe this justifies that the methodologies should not expand to accommodate a more contemporary view, especially uncertainties. The reviewer may not be aware that there is a beta-test version of RESRAD that incorporates Monte Carlo facilities for parameter uncertainties, which indicates an awareness on the part of the developers of the changing methodology.

> 6) I suggest that it is more appropriate to develop SALs by answering the following question: What is the contaminant concentration in soil that results in an acceptable dose limit (for a specified exposure scenario) with a specified level of confidence (given uncertainty in environmental fate/transport and exposure parameters)? I propose use of the equations presented below as a straightforward means of addressing this question.

We believe we have posed this question, along with considerable discussion to guide the reader. What follows this comment is the reviewer's proposed formulation consisting of five equations with some explanation of the notations (which are similar to the ones we have used). We have the following problems with the presentation:

A) It is based, in part, on an erroneous assumption.

B) The introduction of the ratios b_i, it seems to us, clarifies nothing. In particular, if such ratios are to be explicitly introduced, it would be preferable to refer every nuclide to ^{239}Pu; those ratios are available from Krey et al. (1976) and are less awkward in the formulation.

The erroneous assumption consists of the following:

114

c) The **maximum** [our emphasis] total dose due to any individual radionuclide can be calculated using:

$$D_i = T_i \cdot C_i \tag{2}$$

At Rocky Flats, some of the radionuclides are decay products of others; in the most important case, ^{241}Am is a decay product of ^{241}Pu, which in turn decays to ^{237}Np, a long-lived alpha emitter. At present, the levels of ^{241}Am (and ^{237}Np) are rising as ^{241}Pu decays, and they will do so until the early 2030s (Krey et al. 1973; our calculations give the same result). Thus it would be incorrect to assume, for any initial time before 2030, that the proposed equation (2) represents the maximum dose from ^{241}Am and particularly ^{237}Np. Whether or not this would result in palpable error in the total dose remains to be seen from the Task 5 calculations (the early plutonium dose may dominate the much later neptunium dose and render the point moot). Also, different rates of removal of isotopes from the surface soil complicate the question. Our approach was to develop the formulation with sufficient generality that such questions are likely avoided in preference to having them arise later and require additional calculations and explanations.

Krey P., E. Hardy, H. Volchok, L. Toonkel, R. Knuth, and M. Coppes. 1973. Plutonium and Americium Contamination in Rocky Flats Soil. Report HASL-304. U.S. Energy Research and Development Administration, Health and Safety Laboratory.

Reviewer Five

> This reviewer has at least one suggestion for an additional source of information, similar to a computer model, that *RAC* should consider [this seems to refer to the item just below].
>
> P. 29 The candidate computer programs are introduced. The choice of codes for review is sensible but not necessarily complete. *RAC* should at least make a comparison to screening levels *already* calculated for various scenarios by the National Council of Radiation Protection and Measurements (Report 129, issued January 1999, see the reference list).
>
> In addition, a review of how each of these models treats soil ingestion is reviewed in *Health Physics* (Simon 1998) and should be referenced. It can be seen from Table 5 of that publication that soil ingestion values for the GENII code, in particular, are not credible.

The NCRP document (which was also recommended by Reviewer Two) has been examined (it was distributed about the beginning of April and was not available to us during most of the work on Task 2). It will be used to the extent that it is relevant. It is interesting that this reviewer, who elsewhere demands such stringent adherence to the letter of the contract, now advocates that something other than a computer program be examined. Matters related to bringing the GENII database up to date will be dealt with in Task 5.

> In addition, a level of commentary was included in the report which I found to be inappropriate. In particular, those comments directed to the Department of

> Energy, which is neither a sponsor or direct recipient of this report, are out of place.
>
> Furthermore, I found it interesting that *RAC* discouraged the Rocky Flats Citizens Advisory Board (RFCAB) and Rocky Flats Soil Action Level Oversight Panel (RFSALOP) away from the concept of soil action levels. Though I might agree with that insight, I can not help but feel that such advice is inappropriate in this report for the following two reasons: 1) the report is allegedly concerned only with the suitability of a set of specific computer models, and 2) the contract with *RAC* was (apparently) for the purpose of evaluating those computer programs assuming the concept of soil action levels was already accepted. It seems to this reviewer that it is presumptuous on the part of *RAC* to try and steer the Advisory Board and Oversight Panel away from the concept in this document. That level of discussion should be held in public meetings or in contractor/contractee negotiations.

We are confident the reports will be read by the Department of Energy. We consider the recommendations we made to be constructive and entirely appropriate. As to the contractual obligation to comment on and develop soil action levels, we think our report makes it clear that we are fulfilling that obligation. But our proposal made plain our intention also to explore more contemporary approaches to this assessment.

> P.7, 1st paragraph. The text states: "Thus, the same set of soil action levels could be used for determining the need for remediation, planning the remediation and verifying that the remediation has been successful..." It is unclear whether *RAC* is saying that the <u>same</u> soil action level is <u>necessary</u> for all of these activities. There is actually no scientific reason that is apparent to me to force the same action level for all activities. It would be perfectly acceptable and reasonable to have different soil action levels for different activities, depending on their purpose.

We do not understand what the reviewer is objecting to. We had in mind a comprehensive set of soil action levels, based on all relevant scenarios and dose limits. These action levels, after all, do not depend on specific concentrations, and thus they should indeed be suitable for the applications we enumerated.

> P.8, 3rd paragraph. The text discusses the notion that soil action levels are not needed. As mentioned above, this discussion is outside the goal of reviewing computer programs suitable for the purpose intended. It seems to self-defeating as well as a means for the contractor to control the direction of the study, which also seems improper.

Perhaps this reviewer did not have an opportunity to read our proposal. We do not believe there is anything improper in our suggestions for decreasing reliance on soil action levels.

> P. 9. 2nd paragraph. The text states: "In general, we allow both the numerators and the denominators ... to be uncertain quantities." The approach discussed here is appropriate, however, the discussion does not illuminate the fact that <u>spatial</u>

116

> variability is a more important concept to the numerator than is uncertainty (i.e., lack of knowledge).

The statement does not indicate which is more important because we do not yet have final formulations that settle the representation of spatial variability. The reviewer seems confident that this will be the more important component, and that may be the case. But the question is better dealt with in Task 5.

> P.10. Following eq. 2.1-2, it is stated that "...the dose limits are not the same for all scenarios." I don't have a dispute with this statement but it needs clarification. Admittedly, this location in the report is probably not the best place to discuss details of the various scenarios and their dose limits, but it would help to at least reference parenthetically where in the report such a discussion could be found.

Another reviewer suggested saying "the dose limits are not necessarily the same for all scenarios, and this addition may be sufficient to alert the reader. The scenarios sketched in the 1996 DOE/EPA/CDPHE document Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement are not uniform in their limiting doses, and we are allowing for such disparities, but as noted, this is not the place in the text to go into detail

> P. 12. 1st paragraph in Section 2.2. The text states "...The 1000 doses define an empirical distribution..." I have a bit of a quarrel calling this distribution "empirical." Such a term gives the distribution more credibility than it deserves because it implies that the values are derived from experiment or observation. Monte Carlo calculations are only simulations and may not represent reality at all. In fact, this particular distribution characterizes "uncertainty" which is not even a directly measurable quantity. The authors need to better characterize the distribution as a calculation of possible alternatives which include a substantial degree of subjectivity; there is nothing empirical about it.

This usage, in exactly this context, is fairly common, even in authoritative published material (for example, IAEA 1989). In fact, one is doing a kind of "experiment" with a computer, by analogy with taking samples in real world measurements. Throughout the history of Monte Carlo methods (which go back to the 1940s at least), computer scientists regularly described these methods in terms of carrying out experiments with computers. The word "empirical," as the reviewer knows, is intended to distinguish the distribution from its theoretical counterpart. The nature of the process is described in the surrounding text.

> Throughout the report there are a number of locations, where as a reader, I could not determine why *RAC* was discussing a particular subject in detail. The first of these is located on p. 14, 2nd large paragraph. The discussion of the methods for determining weighted breathing rates seems out of place in a major section on Exposure Scenarios. How the weighed breathing rates are determined is best suited for a Methods section (which does not exist in this report) rather than a section which defines the scenarios.

We do not share the reviewer's organizational preferences for the report.

117

> P. 16, Scenario 9. The soil ingestion rate described here (88 grams per year) is an interesting, but not credible, value unless it is an upper bound. First, I cannot help but wonder how a figure of 2 significant digits was arrived at. Second, a continual daily ingestion of 240 mg per day (every day for a year) is not a credible estimate, particularly for adults. There are no studies anywhere, except perhaps those relavent [*sic*] to indigenous populations living primitive lifestyles, that have provided evidence of such high continuous, inadvertent intakes. This particular issue will likely be controversial throughout the entire RFETS evaluation process. Numerous publications in this field should be consulted, e.g., Calabrese et al. (1994), Sheppard (1995), Simon (1998), only to name a few. These references are noted at the end of this review. I note from Table 2.3-1 that similar values have been recommended by *RAC* for additional scenarios and their credibility is equally questionable.

The scenarios proposed and briefly described in the Task 2 report were provided as "examples of the scenarios that are under consideration." An important part of the process has been to involve the panel in the development of the scenarios by thoroughly reviewing studies with a range of possible input values for the parameters such as soil ingestion. We are selecting parameter values for the scenarios using the data from scientific literature for use in developing uncertainty distributions. When data from a number of studies on soil ingestion (Calabrese et al., 1991, Stanek and Calabrese 1995, Thompson and Burmaster, 1991; Simon 1998) are used to develop a distribution of soil ingestion values (with ingestion values for geophasic children removed from the distribution),and with each study weighted equally, then the median, or 50th percentile of the lognormal distribution is 200 mg per day (5th and 95th percentile values of 60 and 730 mg per day).

RAC agrees that most soil ingestion studies, even the more recent studies using a mass-balance approach, are conducted under fairly idealized conditions, or during more mild seasons of the year (Calbabrese et al. 1991; Binder et al. 1986). This timing factor provides conditions where children may have more ready access to open play areas and outdoor activities and adults may be more involved in gardening activities. While these values that are derived from studies conducted from a few days to a few weeks are quite valid in estimating daily soil ingestion rates, there is a need to carefully consider the implications of translating this daily soil ingestion rate to an annual soil ingestion rate when the year includes large periods of time where outdoor inadvertent soil ingestion activities may be somewhat limited by snow cover, frozen ground, and inclement weather. Because we are estimating an annual rate, *RAC* is using the 50th percentile of our distribution of daily soil ingestion rate, rather than the more conservative 95th percentile value. From the daily soil ingestion rate, we then calculate an annual soil ingestion value based on the number of days of exposure. In the scenario noted by the reviewer, we had chosen a central value from the distribution.

RAC is aware of the publications noted by the reviewer and will reference them in the Task 3 report, Inputs and Assumptions. Our approach to selecting input parameter values will be thoroughly described in the Task 3 report.

> P.19, 2nd paragraph. The text states: "Soil action levels are defined in terms of dynamic models..." This statement came as a complete surprise. Furthermore, I can not see that there is any basis for the statement. Soil action levels are actually a value derived from conditions which are assumed to represent a

118

> steady-state contamination condition, an accepted dose standard, and a lifestyle description (which is used to describes the pathways of potential exposure). The only use for a dynamic model would be if the contaminant has to be modeled from its release point until environmental conditions equilibrate or at least, become predictable. However, I would never want to base soil action levels on such calculations. I see no use for this sentence.

Dynamic models *are* the basis for these calculations, and we strong believe this is appropriate. A model of the surface soil compartment, as implemented in RESRAD and other codes, simulates removal of radionuclides from this compartment over time and the movement of the material into ground water (if that option is exercised). It is this dynamic process that gives calculated annual doses that vary with time during the 1000-year period that we are required to consider. The decay chain calculations that run throughout these assessment programs are based on a dynamic model of nuclear transformation. Even when steady-state conditions are applied to estimates, the conceptual (and often the practical) basis for the steady-state is generally a dynamic model represented by a system of ordinary or partial differential equations. To assert that dynamic models are the basis for a calculation does not necessarily imply that transients are being explicitly solved for and examined.

> P. 20. Section 3.1.1 The first mention is made that the temporal scope of the scenarios is 1000 years. If I were to give RFCAB or RFSALOP advice, I would state how ludicrous the idea is of predicting consequences more than 50 years into the future. Not only is there no environmental data or models on which to base those assumptions, human behavior, societal norms, and societal stability, etc. is impossible to predict. Soil action levels should be determined only for those conditions which are presently understood. Anything more than that is part of the "garbage in/garbage out" syndrome of modeling. Furthermore, it deludes the public that scientists are capable of more than is actually possible.

For the record, we stipulate that millennial predictions of the kind required by the contract are, in our opinion, almost meaningless. Even as we carry them out, as we are required to do, we intend to help readers achieve a proper perspective about what (if any) meaning can be derived from such predictions. We would add that in the forecasting business, even 50 years is a very long time.

> P.21. Section 3.1.2 This is a rather small point but the phrase "Figure 3.1.2-2 shows the variation of ^{239}Pu concentrations" should actually read "Figure 3.1.2-2 shows the trend in ^{239}Pu concentrations". It is not incorrect to state that it shows the variation but it is misleading for the following reason, Actinide contamination of soil is extremely variable, primarily because of the particulate nature of most plutonium contamination — a reflection of the circumstances which generated the contamination and its low solubility. Few studies carefully document this variation except on a gross, macroscopic scale. Here the data points are a km apart. Variation of plutonium contamination exists on a spatial scale measured in cm.
>
> Though only a word change is suggested above ('variation' to 'trend'), the idea has greater importance in the discussion which states "RESRAD proceeds on the

> assumption of a uniformly contaminated area..." and "For some scenarios, it could be desirable to subdivide the site .... each having a uniform concentration." What does it mean: "**...could** be desirable? At what spatial scale do you make a determination of "uniform concentration" and what is the rationale for that scale? There is no discussion of the ramification of ignoring the heterogeneity of the contamination, yet, there should be. When spatial variation is properly considered, the extremely wide probability range of possible doses become apparent. It is my opinion that none of the programs reviewed can adequately handle the true spatial variation of actinide contamination in predicting environmental transport and dose to human. Thus, it is necessary to at least state this weakness and possibly discuss the consequences of this inability to model the environment correctly.

(First paragraph) Point taken, but in the text the concern has to do with differences over a two dimensional region, and this seems more appropriately described as "variation." The word "trend" suggests low frequency variation along a line (i.e., one dimension).

(Second paragraph) We think the reviewer knows that this is a question without an easy answer. We are working on it for Task 5, and we cannot answer it in this Task 2 report. The codes reviewed here could be applied to one subplot at a time and the results summed, but the process is complicated to set up and execute and difficult to explain to casual readers, and we are not convinced that such a scheme would be necessary or even useful.

> P.23, 2nd large paragraph. In this paragraph I note that concentration units of pCi per grain are used but elsewhere, units of Bq per gram are used. I advocate two things: 1) SI units exclusively, and 2) consistency throughout the document. Many reviewers give the caveat that they are reporting what previous authors used and thus, are hesitant to change. This negative inertia only serves to continue an outdated system.

This was an oversight. For the illustration cited, the unit can just as well be Bq.

> P.23, last paragraph. The text states: "47 $\mu g\ m^{-3}$ with a standard deviation of 9.0 μm. These units are not stated to be the same though they must be made consistent.

This was a misprint.

> P. 27. Section 3.2 1 found the reference of "introduction of radioactivity into blood through injection" as a contamination pathway to be offensive and inane. It contradicts P. 19 which defines "pathway" to be "the succession of environmental media through which radionuclides move."

Injection of radioactivity has been, for many years, one method of introducing radioisotopes into the body for therapeutic and imaging purposes. This specific intake mode is not likely to be applicable to the problem at hand, but when one is making a generic list of intake modes, this is one of them. Nothing sinister was intended, and we think that would be obvious to any reasonable reader.

120

And in the other matter raised in this remark, we have not confused our usage of the words "mode" and "pathway," as the reviewer seems to allege. A careful reading of the first sentence reveals that the word "pathway" refers back to discussions of pathways (e.g., soil to air) in which some exposure modes (e.g., inhalation) were mentioned. A mode can be talked about in connection with a pathway without being confused with it.

> P. 27, Section 3.2 The speculation that beta emitters in close proximity to the skin may "possibly [cause] skin cancer" should either have a legitimate literature citation that provides evidence of that effect or be removed.

The hedging here had to do with how much, how close, and how long. NCRP Report No. 106 (p. 11) can be cited. [National Council on Radiation Protection and Measurements (NCRP). 1989. *Limit for Exposure to "Hot Particles" on the Skin.* NCRP Report No. 106. NCRP, Bethesda, Maryland.]

> P. 28. The discussion of the various metrics of radiation dose (with its various combinations of weighting factors) seems out of place in a section on "Exposure modes." Furthermore, I doubt whether discussion on the concept of "effective dose" has a place at all in that only the ICRP has found a use for this concept. I have never been convinced that the concept, which simply dilutes the absorbed dose to a specific organ, by the use of weighting factor (less than 1.0), to be of any value. Risk coefficients (other than those derived by ICRP) are organ specific and not applicable to effective dose.

The dose limit is expressed as (annual) effective dose, and we are required to use that metric. We are also required to perform corresponding estimates of risk.

> P. 29 The candidate computer programs are introduced. The choice of codes for review is sensible but not necessarily complete. *RAC* should at least make a comparison to screening levels *already* calculated for various scenarios by the National Council of Radiation Protection and Measurements (Report 129, issued January 1999, see the reference list).
>
> In addition, a review of how each of these models treats soil ingestion is reviewed in *Health Physics* (Simon 1998) and should be referenced. It can be seen from Table 5 of that publication that soil ingestion values for the GENII code, in particular, are not credible.

As noted previously, we will consider NCRP Report No. 129 for its applicability. However, the reviewer needs to be reminded that we were required to consider computer programs, not tables or unprogrammed models. The matter of the GENII predictions may have to do with an obsolescent database, which we will be examining in Task 5.

> P. 43. Mention is made that GENII uses organ weighting factors from ICRP 26 (a 1977 publication). I have to question why such old data is used (newer factors were recommended in 1991 by ICRP) though again, the doubtful usefulness of the effective dose is still an issue. Though this may not be the forum to debate the wisdom of the effective dose concept, it is particularly important that public

> readers understand that actinides do not contaminate or expose the body uniformly, thus, the organ dose to the lung, liver, or skeleton will be greatly diminished through the use of the weighting factor. The unfortunate situation exists that the same metric (Sv) is used for both equivalent and effective dose, thus leaving the uniformed [*sic*] reader with little information as to what the calculated dose really applies to.

Indeed, this is not the forum for debating the usefulness of the effective dose, which we are required to compute. The GENII database will need to be made comparable to that of RESRAD to permit meaningful comparisons, and this is work for Task 5.

> P. 53. Paragraph 5. *RAC* again urges "everyone ... to pay less attention to soil action levels and instead concentrate . . ." Again, it seems inappropriate that the contractor attempts to circumvent the intention of their task in print. This level of discussion should be relegated to workshops and discussion sections.

We strongly disagree that we are attempting to "circumvent the intention of [our] task in print." We fully intend to satisfy the terms of our contract and calculate soil action levels; there has never been a question about that. But we believe that such hazard indices conceal information that ought to be explicitly reviewed, and we intend to remind all parties to the discussion of that fact and to direct their attention to other ways of viewing the relationship between radionuclides in the soil and possible consequences — as we have every right and obligation to do.

> P. 54. The recommendations to the Department of Energy regarding their choice of computer interface is embarrassingly out of place in this text. DOE is neither the sponsor or a recipient of this report. Such recommendations should be make by private communication from the contractor to DOE or at most, brought to light public meetings.

This remark is very much out of place and is contradicted by other reviewers. A careful reading of the recommendations would have indicated that we were not criticizing the choice of an interface or that the graphic user interface (GUI) did not serve a purpose for many users of the program ("We are not suggesting that the GUI be eliminated . . ."), but only that it gets in the way of using RESRAD in the way we want to use it. We pointed out how the program can be made more useful for applications like this one, without changing anything about how most people use it. It is appropriate that such recommendations be conveyed in a context in which the relevant subjects and motivations are under active discussion, and that the recommendations be precisely documented, as they are in this report.

References

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International Atomic Energy Agency (IAEA). 1989. Evaluating the Reliability of Predictions Made Using Environmental Transfer Models. Safety Series 100. International Atomic Energy Agency, Vienna, Austria.

NCRP (National Council on Radiation Protection and Measurements). 1999. Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific

122

Studies. NCRP Report No. 129. Bethesda, MD: National Council on Radiation Protection and Measurements.

Sheppard, S. C. 1995. Parameter values to model the soil ingestion pathway. Environmental Monitoring and Assessment 34(l):27-44.

Simon, S. L. 1998. Soil ingestion by humans: a review of data, history, and etiology with application to risk assessment of radioactively contaminated soil. Health Physics 74(6):647-672.

Reviewer Six

These are useful comments from a very well-informed reviewer. We are particularly impressed by his (or her) examination of background documents. The reviewer's major comment, concerning our view of treating the parameterization of each scenario as a set of constants indicates that we have not yet communicated this part of our methodology clearly, because the comment does not accurately depict our view or intended approach. We do not intend to respond to this point in detail here, but rather we will amplify the discussion in the Task 2 report in an effort to clarify it for readers (or possibly defer some aspects of it to the Task 5 report). If it is not clear to this reviewer, we accept that we probably have not made it clear to anyone.

> P. 7. Points 3 and 4 would benefit by being generalized to encompass dose or risk coefficients, and annual dose <u>or</u> lifetime risk. This would be less parochial (i.e., radiation oriented) and more consistent with Superfund. Soil Action levels are most frequently used for chemicals, based on lifetime risk and the present action levels based on dose are themselves a special case that is derived from the Superfund risk criterion of 10^{-4} lifetime risk from carcinogens (40 CFR Part 300.450(e)(2)(I)(2)).

We do not disagree in principle, but we agreed to the dose criteria as part of the contract. A lifetime risk calculation is required for each of the dose criteria, and we will provide that.

> P. 10. Following eq. 2.1-2: ... are not *necessarily* the same for all scenarios.

The dose limits presented to us are not all the same, but we agree with the added word.

> P. 11, First full para. following eq. 2.1-9: The probability that the inequalities hold in the real world also depends on the accuracy of the scenario choice. The standard must be met for most real world people, and with a reasonably good probability.

This is part of our reason for viewing the scenario as a standard rather than a statement about real people. The standard must be carefully defined with the aim in mind that meeting it would protect most real world people finding themselves in the exposure situation hypothesized by the scenario. There is no difference of opinion on the goal, but only on the best formulation for attaining the goal.

> P. 13. Last para.: Scenarios do not usually represent single people, but significant subgroups of a population that, it is assumed, can be represented by a common set of characteristics. (E.g., it would be inconsistent with the concept of RME individuals to use average breathing rates, unless the RME individuals received above average exposures for reasons not related to inhalation.)

This depends on the use to which the calculations are to be put. And we are not proposing the use of average breathing rates for a scenario subject, as the next paragraph should indicate.

> P. 14. First full para.: Why must this process be any different from that described for environmental parameters in Section 2. 1 ?

In principle, it is not. But as a matter of interpretation, combining the uncertainties associated with the source term and environmental transport with parameter distributions for a conceptual population that may or may not ever contribute a member to the envisioned exposure conditions yields a composite distribution that requires careful probabilistic interpretation, and to us the interpretation seems strained and possibly misleading. We must think of the probability that an individual chosen at random from such a population, given that such an individual encounters the exposure conditions of interest at the specified place and time, receives an annual dose not exceeding the given limit. It seems preferable to us to formulate the scenario according to the principle that the parameters should be chosen to define a hypothetical individual who would experience a dose per unit exposure at least as great as, say, 95% of the population that the individual is assumed to represent. Then this fixed scenario functions as a standard, which can be specified by listing its parameter values (not a set of distributions). With this formulation, our interpretation of the probabilistic statement is simple: it is the probability that the dose limit will not be exceeded for this scenario, period, and we may focus attention on the environmental uncertainties. This formulation is more conservative than the one the reviewer prefers, but we think not unreasonably so. Of course it is possible to combine the two kinds of distributions, but the question is, should one?

> P. 21. Is there any way to provide for the possibility of colloidal transport in the uncertainty analysis?

We are considering this question. We do not yet know the answer.

> Section 3.1.4.2. Table 3.1.5-1 indicates that the dose from Am-241 could be increased by a factor of two if ground water is included in the analysis. Given the major contribution from this isotope, it would seem imperative to include this pathway in calculating soil action levels. This is particularly the case for the rural residential scenario Tier 11 case, when institutional controls are assumed to be absent. It should also apply to any residential case applied to Tier I analyses if institutional control is not assured for the full 1000 years. It seems reasonably obvious to this reviewer that it should be assumed that the RME individual will use ground water if it is not institutionally prevented.

We substantially agree.

> P. 40. The resuspension issue is clearly critical In view of the precedents found in draft Task I for action levels at other sites it would appear to be essential to

124

> make a strong case for any lower value to be applied to the Rocky Flats site. Perhaps an uncertainty analysis of environmental parameters, coupled with a somewhat conservative view on the degree of assurance required for compliance with the standard by the RME individual would be the most supportable approach. In this regard (the degree of conservatism appropriate), to what extent can we predict the effects of climate over a 1,000-yeat period on enhancement of resuspension?

The question is a reasonable one and is similar to one raised by another reviewer. The programs can be manipulated to permit analysis with different assumptions about resuspension, but the only real calibration available to us is tied to measurements made under the environmental conditions of 1983–1984. It is possible, for example, to assume that a tornado (or fire) denudes the soil east of the 903 area and enhances the resuspension for nearby off-site scenarios who may have escaped the immediate fury of the natural events. We can explore such possibilities, but our time and budget will severely limit the extent to which they can be pursued.

> Section 4.3. Could not deterministic comparisons be made, once the relevant values of parameters (e.g., 50 and 90% confidence levels) had been evaluated using RESRAD?

Without making a commitment, we will consider this possibility.

> P. 43. Second full para.: I assume that the outdated external and internal exposure factors in GENII would be updated by *RAC* for the relevant isotopes for any use of this model.

> P. 45. The result showing differences for external exposure is particularly disturbing. This pathway should be the least subject to large differences between models. I would have thought that this code would by now have incorporated the newer calculations of Eckerman and Ryman reported in Federal Guidance Report 11, in place of the old 1981 calculations of Kocher, or the 1983 soil calculations of Kocher and Sjoreen.

To the extent possible, we will reconcile the databases of GENII and RESRAD. Even RESRAD does not have the most up-to-date dosimetric data.

> P. 53. Next to last para.: While I emphatically disagree with the comment that soil action levels will become cumbersome to deal with and will offer little if any advantage, I equally emphatically agree with the suggestion that primary attention should be paid to the dose levels achieved. Even more to the point would be to pay attention to the lifetime risk levels achieved. To this end, it is recommended that the Task 5 report include a calculation of the lifetime risk for each of the action levels. This can be carried out without any difficulty using the tables in Federal Guidance Report 13 - Part I "Health Risks from Low-Level Environmental Exposure to Radionuclides" by Eckerman et al.

We think the reviewer would find that soil action levels for individual radionuclides would become cumbersome if represented by correlated distributions (think of a computer file with

1000 lines, and ten or so numbers to a line). But if isotope ratios derived from measurements by Krey and others may be assumed, it would be possible to maintain a distribution of the SAL for ^{239}Pu, which would be derived with the assumption that the specified isotope ratios prevailed at the starting time for the scenario.

The calculation of the lifetime risks is part of the contract and will be done.

> Regarding the first point, introducing uncertainties should assist rather than deter the selection of action levels. The relative abundances of the various isotopes should not vary widely over the areas of significant contamination, and thus the conditions set forth inequation2.1-1 should be relatively stable across the relevant area at the limiting levels of concentration for each scenario. It should not be difficult to select a single value for each isotope, based on the probability distributions for the SALs (as shown in Fig. 2.2-1), once the desired probability of satisfying the dose criterion is specified. Such values would be clearly easier to implement onsite during cleanup than the implied alternative, which could require extensive inputs of expensive-to-obtain point-by-point analytical data, in addition to field use of computer modeling.

We do not disagree with the comment, if we are interpreting it correctly. We think it likely that the relative abundances estimated by Krey et al. (1976), corrected for radioactive decay and formation of progeny from the early 1970s to the baseline time for the SAL, can be assumed to vary little from point to point. We did not intend to recommend the excessive analysis that would result from ignoring these isotope ratios, but we wanted to leave the handling of the question open until we formulated the Task 5 calculations.

Reviewer Seven

This reviewer's extensive and thoughtful comments deserve a fuller response than we are able to give them.

First, regarding the concern about excluding MEPAS. The rigidly enforced schedule of this project made it unavoidable that computer programs for which access could not acquired in the first two or three months could not be given further consideration. The intent, of course, is not to express prejudice against MEPAS, but we would be unable to treat MEPAS on an equal basis with the other programs. We have said in response to a previous reviewer's comment that we will consider making some deterministic calculations with MEPAS, if there is time to carry them out and include them in the report, but we do not intend that this statement be taken as a commitment that we will do so.

> The draft is thorough, accurate, and credible. It is coherent, and even though there were several authors, it does not appear to be written by a committee. However, it will not be easily understood by those unfamiliar with the task requirements, history of this particular issue at Rocky Flats, etc. Consideration should be given as to whether the final report for each task should also have a separate brief document (not the abstract in the draft) that presents the results and conclusions in a manner more generally accessible to interested non-professionals. More important, if it is not already planned, *Risk Assessment Corporation* (*RAC*) and the Oversight Panel should be planning One or more

126

> summary reports at the end of the project that present the overall conclusions in a manner easily understood by various segments of the public.. This might include audio-visual summaries as well as written ones. (It would probably be more efficient overall if the summary segment on each task was prepared at the same time that the final report on each task is completed).

In technical reports, one is obliged to deal with technical matters in some detail; otherwise, reviewers complain that the authors have not been forthcoming with supporting information. We believe that an executive summary of the final report can deal with the reviewer's concerns, and we take the point about preparing task summaries as the tasks are completed.

> Page 3-4. The distinction between deterministic and probabilistic approaches is presented about as clearly as it could be. However, it should probably be stated that the 1996 soil action levels (SALs) were developed deterministically, and *RAC* might want to provide its opinion as to whether that was standard *at that time,* or whether in *RAC*'s view a probabilistic approach would have been the "contemporary modeling practice" even then.

It would be awkward to try to designate a date marking a transition of contemporary practice in this regard. The development of uncertainty analysis as a part of environmental assessment methodology goes back at least to the 1970s. It still lacks uniform and explicit acceptance by government agencies, particularly where regulatory definitions are involved, but we believe it is fair to say that contemporary practice in assessment methodology supports uncertainty analysis (and has done so for a decade or more).

> Page 4. 1 suggest adding one or more summary tables that provide the key comparative features of the five models considered, either here or in Section 4 (e.g., developer, year first published, applicable directly to radionuclides, yes or no; etc.) Editorial: GENII is termed a "mature and stable" product. No other model is anointed with either such a fulsome (or denigrating) short summary. (RESRAD and MMSOIL probably deserve tile same description.) There should be a summary statement for all or for none.

We will consider the comparison table. "Mature and stable" meant nothing more than that GENII has been through numerous versions and is unlikely to be modified further. But RESRAD is likely to undergo further development; we do not know about MMSOILS.

> Page 5. Editorial. Is it worth considering telegraphing the conclusion regarding previous and current versions of RESRAD here?

Probably so.

> Page 7. Editorial. The statement in the first paragraph "The soil action levels as defined do not depend..." will probably be confusing to many readers. I suggest this paragraph be broken in two, with one paragraph defining soil action levels and a second one, which might come later, discussing the "sum of ratios" topic. Also, perhaps an example could be given to more specifically show the

> relationship of soil action levels to actual concentrations (need for remediation) and the other uses.

We will add another clause to the flagged sentence. We would prefer to defer comparisons of soil action levels with existing levels in RF soils to Task 5.

> Page 10. Editorial. it might be helpful for there to be a second figure, after Figure 2.1-2, to show the geometric interpretation for a slightly more complicated scenario, especially since *RAC* emphasizes the sum of ratios approach throughout the draft. (Also, shouldn't this figure be 2.1-1, to be consistent with later numbering? (See, e.g., Fig. 2-2.1 on page 13).

We do not know what kind of second figure would be effective. A three-dimensional interpretation would be less clear because of the difficulty of indicating the inside, outside, and boundary of the tetrahedron that would correspond to the triangle in Figure 2.1-1 (number corrected), and we do not think such a figure would add any information. Perhaps some words added to the caption, indicating that all combinations of C1, C2 for which the point (C1,C2) lies on the line would make SR = 1 (although the labels in the figure also indicate this).

> Page 12. Editorial. Most readers who get this far will know what Monte Carlo techniques are, but Latin hypercube sampling may be less familiar. Do you really need to mention it specifically, or could you just refer to "other sampling techniques"?

It is not necessary to mention Latin hypercube sampling specifically.

> Page 12 and elsewhere, general point. Intellectually, I understand and agree with *RAC*'s emphasis on the use of uncertainty analysis, though that feature will eventually prove very hard to present to many segments of the public in an educational sense. However, there is another implication. Assuming the original SALs were developed deterministically (and if *RAC* has the view that was wrong at that time -- see my earlier point), then consciously or unconsciously *RAC* is raising the specter that the original SALs should be re-done. This is, as far as I can tell, both beyond the scope of the contract and more important beyond the scope of the agreement between DOE and the Rocky Flats Citizens Advisory Board. *RAC* should not lightly set the stage for such a confrontation. The *technical* answer may lie in the realm of running the models *RAC* chooses (including the newer version of RESRAD) in a "deterministic" manner (using single values instead of distributions, perhaps with a choice of reasonable but high, reasonable but low, and some median level for key parameters), to compare them "head to head" with the original SALs, as well as in the *RAC*-preferred probabilistic manner. This is an important point in my mind, perhaps one of the two most important in my review of the draft.

We do not see the conflict. *RAC* will calculate SALs as required by the contract, but *RAC* made clear in its proposal that its approach was about more than specific computer programs. *RAC* will provide deterministic SALs, along with distributions, and the deterministic versions may or may not agree with the ones that DOE has computed. *RAC*'s methods do indeed imply a critique of the DOE SALs, and we see no way of avoiding this implication of a comparison (but if this

128

document review proves anything, it certainly demonstrates that *RAC*'s methods will also be subject to scrutiny). After reviewing our calculations, DOE may wish to revise its own or it may defend them. It is not up to *RAC* to make decisions about how our information will be used. We do not agree with the conclusion that the deterministic calculation of vintage 1996 must be "wrong" if the uncertainty approach could have been considered contemporary at that time. Assessment analysts have frequently found themselves involved with obsolescent (even obsolete) and new methodologies at the same time. What is new and considered "best" usually languishes for a long time until the nuts and bolts can be assembled to permit everyone to implement it, and sometimes regulatory criteria are not promptly revised to accommodate it. For example, the dose conversion factors in RESRAD belong to a methodology that is at least 25 years old, and the replacement factors from ICRP are now mostly available. But we suspect that the conversion will be some time coming.

> Page 15. The resident rancher scenario has the rancher spending a total of about 15 days per year (one hour per day) off the ranch. I am personally familiar with both ranching and farming families in the northern Rockies and other semi-rural areas, and believe that this underestimates the amount of time spent off the site (trips to town for supplies, coffee shop visits with other ranchers, picking up the mail, longer duration business or family travel, vacations, etc.). Unless the scenario has been accepted by the RSAL already, or *RAC* has studies to support the one hour per day estimate, I <u>recommend</u> increasing it to 2 hours per day, and based on the ranching families I know, even 2 hours is probably conservative (that is, a low estimate of the time spent off the ranch).

> Page 16. The current industrial worker scenario is an excellent addition. If the overall list of scenarios is shortened for some reason, this one should definitely be retained. As a minor point, if the current union contract stipulates only 2 weeks of vacation for a new employee, then 50 weeks is an appropriate time period. However, if there is a pattern of overtime suggesting that 2100 hours per year (or 50 weeks total time per year) is routinely exceeded, even for new employees, then 52 weeks per year should be used. In contrast, if new employees are given more than two weeks vacation per year, and there is no pattern of overtime, then a smaller number of weeks should be used.

While the recommendations made by the reviewer are reasonable for exposure scenarios in a retrospective study, for this project we must develop exposure scenarios for the distant future when we are quite uncertain about the land use. As a result, we think it is appropriate to bias some of the scenario parameters in a way that would increase estimated annual radiation dose. One of these parameters is time spent on site. We are not certain what the future may hold and therefore assume, for some of the scenarios, on-site occupancy time of 52 weeks per year. We are still in the process of finalizing our scenarios and will consider the comments made by the reviewer very carefully.

> Page 20. Editorial. The phrase at the end of the second full paragraph, beginning "sometimes they cannot..." may shed not fight but rather cast a shadow on the first clause. I recommend it be dropped. Alternatively, in later reports on other topics, *RAC* could explicitly point out where it strays from the highly appropriate "general guidelines" that are presented here.

This is only a "full disclosure" impulse that is based on our experience. If we elaborated to explain those occasions when the guidance cannot or should not be followed, it would become tedious. Since we have used the word "try" in the sentence preceding the one in question, we think deleting the offending sentence would be the better choice.

> Page 21. It is appropriate to mention the colloidal transport mechanism. Even though there is no body of data available to calibrate the models for this phenomenon, is there a way for some of the model runs to incorporate "worst case" assumptions as the analysis proceeds? Or perhaps there is another way to deal with this issue in a later task? It is important for *RAC* to try to find a way to address this issue, if at all possible under the terms of the contract it has. At the least, *RAC* should consider providing a perspective on the potential importance of such transport, and/or recommendations how DOE or others should follow up on this issue, either right away or in the near future. Otherwise, at the end of the project, no matter what *RAC*'s overall conclusions are, there will be a lingering worry that this potential threat will dwarf any other potential risks in the future.

We continue to ponder this question. We do not know what would constitute a worst case for colloidal transport, and we are doubtful that much theory can be developed during this project.

> Page 21. Regarding dividing Rocky Flats into smaller plots of land for the purpose of this project, I firmly agree with *RAC*'s "reluctance to recommend this refinement". In the final version of the report, I suggest that *RAC* be even more conclusive. This could mean a firm opinion that this degree of refinement is simply not justified, given known site conditions (in particular, the small area of high contamination, which will no doubt dominate the results), or, less satisfactory in my view, listing the "factors" that, after "careful evaluation", would require such a step, and then concluding the evaluation means this step not be taken.

This issue affects calibration of the resuspension model as well as routine calculations, and the full solution will have to await Task 5. The problem will be better formulated in terms of how the soil concentrations should be spatially averaged.

> Page 26. Editorial primarily, with one substantive suggestion. Section 3.1.4.2 states that the *RAC* team agrees with the cited 1996 study, but then states that research should be continued and groundwater issues should not be dismissed. Colloidal transport could well be mentioned as a specific research/monitoring need that others should definitely pursue (see my earlier comment), and would give some precision to the statement, In addition, one of the scenarios postulates groundwater use, and could be mentioned here as one step *RAC* is taking to deal with groundwater. In that regard, as a suggestion, some consideration could be given to revising one of the Woman Creek scenarios to substitute ground water in whole or in part for surface water. However, I do not recommend that additional scenarios be added-there are enough already.

> Page 26. I am not certain the phrase "simple screening exercise" does justice to the choice made and analysis done by *RAC* and the way both are presented.

130

> Instead, I suggest that *RAC* not use that phrase and elaborate more on why it chose to do what it did and reached the final view that it did ("should perhaps be investigated further.")
>
> Page 26-27. Primarily editorial. The last full paragraph on page 26 and the next paragraph on page 26-27 should be clarified and firmed up. One change is to move the sentence starting "For the radionuclide..." up to be the last sentence in the prior paragraph, and starting the next paragraph with "The results of this exercise... " The implications of Table 3.1.5-1 should probably be spelled out more explicitly. Even more important, there should be a better explanation of why *RAC* "will ignore the groundwater pathway" (in fact, one of the scenarios includes it), and what the implications are (minor, major or unknown) of ignoring it. In addition to its technical implications, the way these two paragraphs ate worded raise the same specter noted earlier regarding colloidal transport. I can imagine the reaction of some segments of the public: How can we put any trust in the *RAC* conclusions if, according to their own report, *RAC* chose to 'ignore the groundwater pathway"?

We have incorporated these suggestions for editorial changes and have added some additional text to provide further explanation of the Soil Action Levels that include the groundwater pathway. In doing so, we have uncovered several misinterpretations of the analysis and have made corrections.

On the basis of these comments and the fact that one of the scenarios included groundwater ingestion, we have decided to include the groundwater pathway in our calculations for at least one of the scenarios. The groundwater analysis will only consider dissolved phase transport because colloidal transport models have not been extensively developed and could not be implemented within the time and budget constraints of this project. We note that this will probably make little difference in the overall action levels because doses are driven by inhalation and external radiation sources for most nuclides. The nuclides where differences are expected include ^{241}Pu, ^{241}Am, and ^{234}U.

> Page 29. I have two major comments on this page.
>
> First, the draft states that RESRAD was included "in accordance with the contract," which is of course true and also fundamentally needed-since this project is the direct result of the earlier use (of an earlier version) of RESRAD that led to the levels currently embodied in the cleanup effort. However, the use of the quoted phrase implies that but for the contract, *RAC* would not have chosen RESRAD. In short, this is damning with faint praise. Is this what *RAC* believes? In other words, on the basis *of* the five criteria, would RESRAD have been *rejected?* If so, say so. If not, and RESRAD would on the merits meet the five selection criteria (I think it definitely would), say so. (Editorial: why is "nominal" used before "criteria"? Are there "nominal" criteria and separate "really important" criteria?)
>
> Second, the fifth criterion sets the final stage for rejection of MEPAS, though the scenery for this final act was put in place earlier in the draft report. I take at face

> value *RAC*'s statement that the criteria were developed before final decisions were made, and I understand the practical reasons MEPAS was dropped (presented on page 4 1). However, this is not totally satisfying. MEPAS is very well-known in the modeling community, as indicated by the benchmarking exercise cited in the draft, and at least in my experience is for more widely known (and understood-and used) than *GENII. (GENII* was not included in the benchmarking exercise.) In my opinion, it is a very serious matter that MEPAS was rejected, even though I understand why (because the source code was not provided).
>
> Separately, as part of this review, as a policy issue, I am recommending that the Oversight Panel consider formally asking DOE to direct Battelle to release the source code immediately for *RAC*'s evaluation, even if on a confidential basis. In a more technical mode, for *RAC*'s consideration, I strongly urge that *RAC* determine if there is a way that MEPAS can be evaluated, even though (and if) source code or the equivalent "special instructions" (page 41) is not available to you. One possibility would be to reduce the results *of* the probabilistic runs *RAC* makes to single or a small set of single values (such as mean, median, mean + one standard deviation, mean - one standard deviation) and use these as inputs to a few runs of MEPAS. There may be other approaches that skilled modelers can conceive that would overcome the problem that the "front end" of MEPAS as now available to *RAC* does not lend itself to the use of the Monte Carlo approach that *RAC* understandably prefers. (In fact, it seems likely to me that this particular problem has probably been faced conceptually in recent years as the probabilistic approach has become the preferred approach, while many earlier models, not just the ones *RAC* is considering, were developed based on a "deterministic" basis.)

It is virtually certain that RESRAD would have been included in the lineup in any case, and perhaps the language used here should clarify that. The word "nominal" refers to the fact that these criteria were stated in the RFP and proposal, but other sections of the draft report indicate why some of them (e.g., (2)) should not be interpreted too literally (pure validation results are unlikely to be available for the codes, for reasons indicated in Sections 4.1.1 and 4.1.2, but they may be suitable for some validation comparisons using local data). We can drop the word "nominal" if it causes confusion.

As to MEPAS, insufficient time and resources are available at this point in the project to prepare front-end code for doing uncertainty calculations with MEPAS. We hope the panel will not follow the reviewer's well-intended recommendation to make another attempt. We have indicated previously that we will consider performing some deterministic calculations with MEPAS for Task 5 if time and resources permit, although we cannot make a firm commitment to do this.

> Page 31. Editorial. Many readers will not automatically understand that "claiming validation is akin to accepting a null hypothesis." Perhaps a better comparison can be found.

We do not know a better analogy. Perhaps more explanation could replace the reference to a null hypothesis.

137

> Page 33ff. The issues related to different versions of RESRAD, different manuals, etc. are as well presented as they possibly could be. However, I recommend *RAC* consider, either in this report or perhaps better in a later report, presenting in some way (perhaps using tables) major differences that would result if the newest version of RESRAD were run, compared to the version used to develop the original soil action levels. My own prediction is that except for the soil resuspension issue, there will probably not be dramatic differences. If *RAC* does not undertake this comparison as part of its original work, some entities, including very possibly the Oversight Panel itself, will ask that it be done later.

We will show the comparison in Task 5. The differences are all in the resuspension pathway, and if that is exempted from the comparison, there should be no difference.

> Page 36. Editorial. Why is "virtually" used before "exhaustive"?

Clients and reviewers will always find something else that they want to see in a printout.

> Page 37. *RAC*'s recommendation that DOE provide the RESRAD source code more readily is right on the money, and separately I am recommending that the Oversight Panel itself make that recommendation to DOE. If I understand the draft correctly, *RAC* itself is able to resolve the problem of the inconsistencies in the materials and can work with the source code available to it. Instead, the spirit of *RAC*'s observation is more to advance the quality of RESRAD in the long term, not to solve a current need that *RAC* has.

Contrast this with the inexplicably negative comments of another reviewer concerning this recommendation.

> Page 37. Editorial. I suggest adding the word "regarding" between "have" and "unauthorized".

This was a misprint and will be corrected.

> Page 38. Editorial. I suggest that for clarity, "(AF)" be added after *area factor*.

We will do this.

> Page 37-41. This was a particularly hard section to understand. Perhaps the easiest solution is to present part of the overall conclusion that begins on the bottom of page 40 ("In general ....) early in this paragraph, as a roadmap for the entire section. An additional idea might be to break this into smaller subsections. Because of the overall importance of the resuspension issue, this entire subsection should be made crystal clear. This is the only subsection that needs substantial editorial work to improve its clarity.

We doubt that we can make this material crystal clear for the casual reader, but we can add some prefatory material, as the reviewer suggests. The subject is technical, as is the RESRAD supplementary document that details and defends the changes. We do not think that several smaller technical subsections would be clearer than the one larger technical subsection. Without undertaking a rather long textbook type of exposition of the substantial body of theory on which this material depends, we really do not know how to make it clearer to a general reader. We certainly can flag the details as being of primary interest to specialists (as we did for the equations defining SALs in Section 2.1) and rely on the prefatory summary to give the general reader a qualitative idea of what the results are.

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MEMO

May 26, 1999

From: Dr. John Till
To: Ms. Carla Sanda

Carla, attached are responses to the panel's comments on the Task 2 report for distribution.

Thank you.

John Till

Responses to Panel Comments on RSAL Task 2 Report

We repeat the reviewer's comment then follow it with an indented response. In some cases, the comments do not question the Task 2 report but make a general statement that does not require an answer.

Mary Harlow

While I am pleased by the overall direction of the study, I am concerned as to whether the scope of work as outlined in the RFP is being met. Specifically is *RAC* looking at the interim RSALs and reviewing their development and the input data used to set them? The scenarios used to set the interim SAL's must also be reviewed as part of this process.

> We have carefully reviewed the scenarios used in the interim RSALs and made the decision, with the Panel's approval, to use those scenarios along with four additional scenarios that *RAC* developed. In addition, we are providing commentary on some of the parameters, models, and approaches that were used in the DOE/CDPHE/EPA RSALs as they pertain to implementing our approach. As discussed, we do not intend to "critique" every element of the previous RSAL calculations; rather, we do plan to explain where there are differences and why we have chosen one method above another.

<u>The report is difficult to read and follow.</u> Paragraphs in the report need to be broken up by double spacing and shortened where possible. Page 24 is especially tedious to read and long. Isn't there some way to break out topic areas to give the reader some ideas as to what the page covers? Consider using sub headings. Each section should have a summary paragraph at the end.

> We will give careful thought to making the report as reader friendly as possible. This is a very technical report and it is quite important to have the level of scientific and mathematical detail so other scientists have sufficient information to critique our work. However, we will try to provide some type of summary information for nontechnical readers.

Change title to RADIONUCLIDE SOIL ACTION LEVEL OVERSIGHT PANEL.

> We will do this

Offsite impacts and how they could or should be considered in selecting a model are not discussed. This is part of the scope of work. The goal of the project is to protect people who may in the near or distant future come into contact with a site where radionuclides contaminate the soil at levels above background and to also look at offsite impacts.

> We are very aware of the concern about the future impact of groundwater and surface water pathways and are examining a conservative calculation to address the question of whether the groundwater pathway can be ruled out of the current analysis. We understand the importance of groundwater and surface water pathways in the long-term and include the groundwater pathway in one of our scenarios.

We do recognize, however, that our assessment of the groundwater pathway is limited by the complexity of the pathway. More importantly, in the current analysis we have developed conservative scenarios on the premise that if the onsite scenarios are protected, then others onsite and offsite will be protected in the near and distant future.

Page 8 of the draft includes a discussion on the avoidance of soil action levels altogether and to base remediation planning and verification on direct simulations with the data, models and scenario definitions that would have been used to calculate the soil action levels. The task is to review models and specifically to look at other models and determine whether they are applicable to RFETS.

We are reviewing the models for their applicablity to the RFETS. As we stated in our responses to the peer reviewer comments, we will calculate soil action levels as required by the contract, but we think that our approach is about more than specific computer programs. We will provide deterministic soil action levels, along with distributions, and the deterministic versions may or may not agree with the ones DOE has computed. The discussion of avoiding soil action levels altogether is a point of caution of the inherent weaknesses of computer modeling and the strict reliance that we often place upon them.

Page 3 of the Peer review comments discusses a maintenance worker scenario that would take care of the grounds. Vegetation management will be necessary at the site. Please comment on this scenario.

This scenario would represent a person who spends a good portion of time outside working around the site; however, this person proposed by one of the peer reviewers would not live onsite. Meantime, we developed the rancher scenario as a person who spends time outside working in the garden *and* lives onsite year round.

Please provide information as to when *RAC* plans to review the INPUTS AND ASSUMPTIONS, and the methodology used to calculate the current interim RSALs. The panel needs to have an opinion on the original process and how the RSALs were originated. If the original methodology is not evaluated for strengths and weaknesses, it will be very difficult for the RSALOP to recommend an alternative approach to calculating RSALs. At what point in the review will this be done and documented?

We are on track with the task report schedule. The draft Task 3 report on Inputs and Assumptions will be available on July 8.

RAC did not discuss the various models' capabilities to address offsite exposures. This was requested in the Scope of Work. Please include a discussion on each model's capability to model offsite exposure.

We have addressed this in our responses to the peer reviewer comments.

The report needs to provide backup information supporting the choice of only one of the two models considered. It is important that we have defensible, hard evidence to explain the choice *RAC* has made in regards to models.

We have addressed this in our responses to the peer reviewer comments.

Critical testing with real site data will be necessary to substantiate conclusions on appropriateness of models and methods chosen.

Testing models with real site data is problematic. We are using site specific data in our calculations, which at least should make the results fit more closely with what really exists.

Deterministic Versus Stochastic Approach—Several peer reviewers' comments, as well as those from some panel members, have questioned why a deterministic approach as well as a stochastic approach would not be appropriate when determining RSALs.

This has been a topic of great importance during the past months of the project. We agree that it would be helpful to be able to use both approaches in the RSAL work, but the tight time schedule and our resources demand that we focus on other critical aspects of the project first. As we stated in our responses to the peer review comments on Task 2, we established the scenario according to the principle that the parameters should be chosen to define a hypothetical individual who would experience a dose per unit exposure at least as great as, say 95%, of the population that the individual is assumed to represent. This fixed scenario functions as a standard, which can be specified by listing its parameter values (not a set of distributions).

Monte Carlo calculations represent randomness. Running scenarios with deterministic numbers would provide some comparisons with the original SAL numbers and should be done.

See response above.

Page 24, Page 27. Groundwater and surface water transport. *RAC* states that they will examine the ramifications of dismissing the groundwater and surface water pathways in the assessment and also that they will ignore the groundwater pathway. This is an important pathway, especially since water is becoming more precious as time goes on. We should assume that it is very likely that sometime in the future there will be an attempt made to access the groundwater on site. Please discuss the ability of each of the models to address the water pathway. I would like the surface and groundwater pathway included in this study.

It is not possible to address groundwater to the degree that it should be discussed because of the budget, schedule, and its complexity. We do intend to address these issues in a simplistic manner using models built into the RESRAD code.

Page 26, paragraph 2, should be written to state: "Walnut Creek does not flow into Great Western Reservoir. It is currently diverted around the Reservoir and the flows from Woman Creek do not flow into Stanley Lake. They flow into Woman Creek Reservoir." Neither stream enters reservoirs.

RAC will reword this paragraph; in this section of the report; we were discussing the natural flow of the onsite creeks. We will add a statement that as of 1992, Walnut Creek, which previously flowed into the GWR, was diverted around GWR and by 1996, Woman Creek no longer flowed from the site directly into Standley Lake.

Section 3.1.4.2, page 18 should also be corrected: discharges to surface water do not flow to drinking water reservoirs.

Again, we will modify this sentence to reflect the current drainage patterns at Rocky Flats.

Page 35 First Paragraph, last sentence states "we also recommend enforcement of better quality control for the binding of the document: the pages of the copy we received are separating from the spine and falling out." This statement should be removed, as it is not part of the process. It does not fit in this technical review document even though it is an aggravation.

RAC will modify this statement.

Page 53 Conclusions, paragraph 5 states that everyone concerned with the assessment pay less attention to soil action levels and instead concentrate on the relationship between particular measure or hypothetical sets of radionuclide concentrations in soil and the predicated maximum annual dose to each scenario. Although I think this is an important statement it does not coincide with the RFP Scope of Work, which calls for a review of the interim soil action levels.

We believe the statement is precautionary and correct. It again raises caution to the importance of considering more than simply the soil action levels.

Victor Holm

While these comments are directed to the draft Task 2 Report, I will also be referring to the presentation on scenarios given by Kathleen Meyer and Jill Weber at the RSALOP meeting on April 8.

As I indicated in my letter to Kathleen Meyer on March 10, overall I believe *RAC* is on course and doing an excellent job. I particularly liked the discussion on Soil Action Levels (Sec. 2) and the Site Conceptual Model (Sec. 3). 1 am now familiar with the operation of three of the proposed computer models, RESRAD, GENII and D&D, and I concur that RESRAD is the best choice, I recently talked with Charlie Yu (April 13), developer of RESRAD, and I now have a much better understanding of the pitfalls with the air modeling. I look forward to your presentation on exactly how you will handle air modeling. In addition to the EPA Rapid Assessment Model you may also wish to look at the ICS-3 air dispersion model to see if it can be coded into RESRAD, In addition a beta version of RESRAD-OFFSITE is now available. This tool might be helpful in evaluating offsite exposure even if it can not be formally used because it not finalized.

> We are familiar with ISC 3, and it could be used in conjunction with RESRAD, although our current work plan does not include it.

Although the rest of this letter takes some exception with the scenarios; suggested and the parameters used within them, I wish to assure you that the questions are asked in a constrictive manner I respect the work you am doing and realize that these art difficult questions, I also wish to assure the rest of the panel, although it may seem that 1 am always pressuring for a less conservative standard it is only because the other point of view is so ably represented. We are trying to obtain the best cleanup possible with the limited funds and time available. Whether we agree or not; when the money runs out DOE will build a fence around the site and we will have to live with the results. If this panel ran not scientifically defend the results from what could be a concerted effort to discredit the work then we will have accomplished nothing.

The report discuses nine scenarios. At the last meeting the number was reduced to seven. Three of these are the RFCA scenarios which will not be modified. I do not consider the RFCA scenarios of much use to this study other than as points of reference, The current onsite worker scenario is interesting, but, I fail see how it can be used to set cleanup levels after closure of the plant and the current workers are gone. The infant and child scenarios are useful additions but are unlikely to be the controlling scenarios, We are then left with only one scenario, the rancher, which in my opinion will be difficult to defend because, it not the best or most likely use of the land.

There is broad consensus both among stakeholders and local governments that the site should be used as open space. The EPA, under CERCLA and the NRC, under the License Termination Regulations, both specify that regardless of the intended land use the site must be cleaned up to unrestricted standards unless it can be demonstrated that "complying with the unrestricted use criterion would be prohibitively expensive result in net public harm or not be technically feasible" (10CFR Part 20.1402(d)). The baseline scenarios must then address the unrestricted use standard of 15 mrem. The rancher scenario should be one of these. In my opinion the other should be a suburban resident since this is the most likely unrestricted scenario. These scenarios in no way interfere with the desire of the stakeholders and local government for open space, Since actual land use decisions made by local governments do not necessarily determine the scenario to be used in the cleanup. It is possible that an unrestricted cleanup will not be possible so we also need to consider restricted scenarios, I recommend that the current site worker be used for this purpose. This scenario could apply to an outdoor park worker maintaining vegetation, repairing trails and guiding visitors etc. Since be would work outside on site full time be would undoubtedly have more exposure than the open space user.

My main confusion about the scenarios, which I believe is shared by others, is: Are they in fact standards? My reading of the applicable guidance is that this is how scenarios are normally considered in dose studies. If they are standards, then like any standard, the behavioral variables should be widely agreed upon and should not be site specific. There are many sources for this information; the EPA Exposure Factor Handbook, the NRC guidance, the default values given in the computer program documentation and the open literature. I question how much we should deviate from these sources. Another approach, which some panel members prefer, is to treat them as uncertainty values and use an appropriate probability distribution instead of considering them standards. It appears to me you trying to use both approaches at the same time. You call them standards but, you derived them from probability distributions and then choose the 95th percentile, Perhaps I am being overly concerned about a trivial problem. In qualitative risk assessment the output distribution is supposed to be a measure of the uncertainty in the dose derived from a set contamination level. If the mean of the input distributions are already biased to include a large safety factor will we have an output distribution that is related to actual dose; or, one that is biased. How will we evaluate the extent of the bias?

> We do not feel that the scenarios are unrealistically biased on the conservative side. Our responses to the peer reviewers and our discussion at the RSALs meeting in May helped to further clarify this issue. . We have developed the parameter values for the scenarios according to the principle that the parameters should be chosen to define a hypothetical individual who would experience a dose per unit exposure at some specified level of the population that the individual is assumed to represent." The interaction with the panel on this issue at the May meeting seemd to resolve and complete the discussion on this issue with the panel's support.

This bias is exhibited in nearly all the variables including: hours on site, breathing rate, vegetable ingestion and soil ingestion, From a practical point of view it is not a problem for breathing rate since the distribution used has little relative uncertainty, the mean and the 95th percentile vary by less than 10%. For the child soil ingestion rate the difference is significant It can be argued that the distribution shown at the meeting represents two populations; a normal distribution and a pear uniform distribution. The normal portion represents the uncertainty in ordinary children, while the uniform distribution is probably made up of children with a soil eating condition. The resulting joint distribution shown may not represent the uncertainty of soil ingestion at all. Moreover it is arbitrary and of debatable use to try to select the 95th percentile of a mixed population distribution such as this. One of the concerns some of us have had about this study from the beginning is that excessive safety factors would be introduced into the input parameters during the analysis and then another safety factor would be applied on the results. This was one reason that a probabilistic approach was adapted. If the input distributions are to be biased in favor of conservatism then the entire reason for this approach in questionable, I believe *RAC* needs to explain to the panel what it's approach to safety factors is going to be.

> At the May 1999 RSALs meeting, we discussed our revised approach to selecting soil ingestion values for the scenarios. Most soil ingestion studies are conducted under fairly idealized conditions or during more mild seasons of the year, and researchers tend to point this out in their reports. This timing factor provides conditions where children may have more ready access to open play areas and outdoor

activities and adults are more involved in gardening activities. While these values that are derived from studies conducted from a few days to a few weeks are quite valid in estimating daily soil ingestion rates, there is a need to carefully consider the implications of translating this daily soil ingestion rate to an annual soil ingestion rate. When converting this rate to an annual intake, care must be given because the year includes large periods of time where outdoor inadvertent soil ingestion activities may be somewhat limited by snow cover, frozen ground, and inclement weather. For these reasons, we are using the 50th percentile of our distribution for our daily soil ingestion rate. From the daily soil ingestion rate, we then calculate an annual soil ingestion value based on the number of days of exposure. We think that this is a conservative but realistic approach.

There are several other scenario variables that I recommend be reevaluated:

A. Time on site for the child of 8760 hr/yr does not consider time at school, play with other child or trips and vacations; is this reasonable? Why was the value of 5800 hr/yr in the draft report discarded?

Because we do not know what the distant future will bring, we think that it is appropriate to maximize the time onsite for some of the scenarios. Knowing the current lifestyle of some current farmers or ranchers, it is not totally unrealistic to think they may not be onsite the entire year. Perhaps we could have specified 1 hour or 2 a week for offsite activities; however, we think that if our theoretical rancher is protected, there will be no doubt that others will be protected as well.

B. Time on site for the rancher of 8670 hr/yr does not consider time spent shopping or just socializing with neighbors or vacations. What was wrong with 8400 hr/yr which one reviewer already considered high.

See our response above.

C. Expecting the dry, rocky marginal land at Rocky Flats to provide all the plant food for the entire year is not defendable even at the 95th percentile and it is not the custom on other ranches in Colorado or elsewhere for that matter. Would not 25% be more reasonable?

Again to ensure that future populations will be protected we assumed that all vegetables would be gown onsite. Although the environs may not currently be used in this way, in the distant future some may find it necessary to rely on their garden and other crops and through canning and other food preservation methods use them as food all year.

D. At the April meeting distributions for breathing rate and soil ingestion were shown for the child scenario. The breathing rate distribution is not just a distribution of uncertainty; but, has a strong positive correlation with age, The highest rates correspond to older children. The soil ingestion distribution presumably has a strong negative correlation with age. In fact my reading of the available papers indicates that most of the children with the soil eating condition are less than 5 years of ago. I could find no example in the literature that suggested the. condition is common in teenagers. It is likely that the joint probability of a child breathing more than 8600 cu m/yr and ingesting more than I gram of soil per day in much loss than the 5% you indicated, in fact I would suggest that they are mutually exclusive.

At the May 1999 RSALs meeting, *RAC* presented the final scenarios and discussed our revised approach to selecting soil ingestion values for the scenarios. Also, see responses above.

I have one editorial comments; on p,23 second paragraph I believe the East Gate referred to is not the same as the present East Gate on Indiana St.

The reviewer is correct and we will clarify this statement so it is clear where the measurements were made.

Again I wish to commend you on the generally good job you are doing. I look forward to a continuing dialog, I for one learning a great deal from this project.

LeRoy Moore

One of the peer reviewers for the independent assessment of the Rocky Flats RSALs states that the RSALs as adopted misapply the concept of "institutional controls" in relation to the 15/85 mrem/year dose (see attached "Review Comments on the March 1999 Draft Report for Task 2: Computer Models," section 1, "Application of the 85 mrem/y criterion"), This suggests that the Rocky Flats RSALs violate CERCLA in the way the "institutional controls" concept is employed. What corrections need to be made?

> *RAC* has taken an independent approach to establishing RSALs and has not felt constrained by current regulations. While we are providing commentary on some of the parameters, models, and approaches that were used in the DOE/CDPHE/EPA RSALs, we cannot critique their approach in detail.

Review Comments on the March 1999 Draft Report by the Risk Assessment Corporation (*RAC*) for Task 2: Computer Models

This is a carefully prepared and mostly excellent draft. Prior to commenting on the Task 2 Report, this reviewer reviewed several background documents: the DOE report "Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement - Final, Oct. 31, 1996, and its accompanying "Responsiveness Summary;" the RAS draft report for Task 1, Feb. 1999; the report by Joseph Goldfield entitled 'Breathing Rates of Exposed Persons Residing on Plutonium Contaminated Soil for Calculating Health Effects;" and two papers by LeRoy Moore entitled "Action Levels for Radionuclides in Soils for Cleanup of Rocky Flats" and "Seven Reasons for an Independent Review of the Rocky Flats Action Levels." Review of the first of these reports raised a number of concerns regarding the assumptions underlying their application of the 15/85 mrem/y dose criteria and their choice of exposure scenarios for implementing those criteria via soil action levels, including the selection of parameters characterizing the environment and individuals exposed. I was pleased to find that the authors of the Task 2 draft report reflected many of the same concerns.

By way of background for comments on the Task 2 report, the following summarizes my concerns with the DOE report:

1. Application of the 85 mrem/y criterion.

There is a conspicuous absence of a clear statement of the limited use of the 85 mrem/y criterion intended by EPA, and a strong implication that it is being misused. This criterion was proposed by EPA as an 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, 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 land uses in the distant future for situations in which institutional controls are assumed to -no longer exist. To the contrary, CERCLA regulations require the lead agency to review the efficacy of institutional controls no less often than every five years for as long as they are required to maintain conformance with the level permitting unrestricted use (in this case 15 mrem/y)(see 40 CFR Part 300.430(f)(4)(ii)). 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 under the above periodic review requirement.

It is not obvious to this reviewer, especially for the two types of buffer areas (these are not differentiated in the DOE report), but also for the industrial area, that either the commitments or assurances of effectiveness for the necessary institutional controls exist. The DOE report depends on the documents "Action Levels and Standards Framework for Surface Water, Ground Water, and Soils " (ALF) and the "Rocky Flats Vision." These documents, as well as the "Rocky Flats Cleanup Agreement" (RFCA) and proposed "modifications to the Action Levels and Standards Framework" were not available for this review. However, a "vision" is not a legal commitment, and the discussion of near and intermediate term land uses and, more significantly, the absence of any discussion of long-term land use in the last paragraph on p. 6-15 of the DOE report creates the impression that the state of commitments for and assurances of effectiveness of institutional controls in the future is very uncertain.

The implication of the above, given the long-term contamination present at Rocky Flats, is clear, If the lead agency (DOE), State, and local officials cannot commit to and provide reasonable assurance of maintaining necessary institutional controls in an effective manner for 1000 years, then consideration must be given to cleanup of the site now to levels that would meet 15 mrem/y in the absence of such controls. Obviously, this point is critical to choosing the Tier I Action Levels for the so-called "buffer" and industrial areas.

There is also a need to develop a Tier I level applicable *outside* the buffer areas, since these locations must meet the 15 mrem/y criterion under unrestricted use (presumably under a rural or rancher residential scenario), and the action levels for the immediately adjacent buffer area, at least under the current proposal, would permit significantly higher levels. As noted above, if the necessary assurances for long-term institutional control cannot be met for the buffer and/or industrial areas, this level should apply there also.

2. Exposure Scenarios:

Under CERCLA, the choice of exposure scenarios is intended to assure protection of the "*Reasonably Maximum Exposed*" (RME) individual. This is not the same as the average member of the affected population, nor is it the *most* exposed individual. EPA has devoted considerable effort to clarifying this admittedly elusive concept. The following quotes are typical of EPA guidance:

"...actions at Superfund sites should be based on an estimate of the reasonable maximum exposure (RUE) expected to occur tinder 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.., The intent of the RME is to estimate a conservative exposure case (i.e., well above the average) that is still within the range of possible exposures." ("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 900 percentile of the actual (either measured or estimated) distribution The conceptual range is not meant to precisely define the limits of this descriptor, but should be used by the assessor as a target range for characterizing "high-end" risk " ("Guidance on Risk Characterization for Risk Managers and Risk Assessors," Memo from F. Henry Habicht 11, Deputy Administrator, EPA, to Assistant Administrators and Regional Administrators, February 26, 1992.

cannot address such an offsite scenario, and therefore, GENII has an advantage as a model that may provide dose estimates to offsite individuals. GENII also considers an onsite groundwater pathway like RESRAD does.

MMSOILS

The EPA's Office of Research and Development developed MMSOILS for screening purposes to estimate human exposure and health risk from chemically contaminated hazardous waste sites. MMSOILS simulates chemical transport in the atmosphere, soil, surface water, groundwater, and the food chain and contains a Monte Carlo mechanism for propagating parameter uncertainties into estimates of exposure and risk. It is possible to apply MMSOILS to radionuclides in the soil, but the program has no complete mechanism for dealing with the decay of radioactive materials. Although we included MMSOILS in the list of programs to be considered, we ruled out its use in developing soil action levels for the Rocky Flats site, given the time constraints of this project.

DandD

The computer program, *Decontamination and Decommissioning* (DandD), was designed by the U.S. Nuclear Regulatory Commission (NRC) as a screening level analysis program to provide a simplified estimate of the dose to an average member of a screening group of people. The program gives a conservative estimate that is not designed to be used as an estimate of actual dose. The DandD code includes four exposure scenarios: building renovation, building occupancy, drinking water, and residential. For the residential scenario, the pathways included are external exposure, inhalation, drinking water ingestion, ingestion of food grown from irrigated water, land-based food ingestion, soil ingestion, and fish ingestion. However, the pathways are hard-wired into the scenarios and can only be removed from consideration by zeroing the annual intake of any given product.

A drawback to the use of DandD in the current project is that it is still in its first version and has not been used extensively yet. The documentation that accompanies the code has not been published, nor has the source code been released. This makes it difficult to use the code and even more difficult to make confident statements about how the code functions. Without the appropriate documentation, we could not consider the DandD code further for this project at this time.

In summary, based on the extensive evaluation of the available computer codes carried out in Task 2, we concluded that either RESRAD or GENII could be adapted for the purposes of the project. DandD cannot be counted on in the time available for this project because it is still in an early development stage. DandD is also focused more on screening calculations that makes it less suitable for this project. MEPAS and MMSOILS were ruled out on other practical grounds.

RESRAD

The U.S. Department of Energy (DOE) and Argonne National Laboratory (ANL) developed the computer program RESRAD (RESidual RADioactivity) for the purpose of performing calculations related to meeting the Department's criteria for residual radioactivity. The exposure pathways available include inhalation, external gamma radiation from soil and airborne radioactivity, soil ingestion, drinking water, ingestion of vegetables, meat, and milk, and these can be individually switched on or off to permit the treatment of a variety of scenarios. The original program from 1989 used site-specific guidelines (called soil action levels in this report) based on DOE guidelines. We have used the most recent version of RESRAD (Version 5.82), which differs in some ways from older versions that are still in use.

The main difference in the newer version from the version of RESRAD that DOE, EPA, and CDPHE used in preparing the existing action levels document is how the program treats the resuspension of soil. Given the importance of resuspension in the Rocky Flats area, these changes may be significant. The changes involve the calculation of the area factor (or enhancement factor), which is a factor that accounts for the dilution of locally contaminated airborne dust by uncontaminated dust resuspended from outside the contaminated area. The Task 3 report provides a detailed look at how resuspension is being addressed.

MEPAS

The Multimedia Environmental Pollutant Assessment System (MEPAS) was developed at Pacific Northwest Laboratory under DOE sponsorship. Offered as a commercial product by Battelle Memorial Institute under a technology-transfer agreement with DOE, MEPAS has applications for both chemical and radioactive pollutants, with built-in computation of human health risk. MEPAS includes air transport models in addition to surface water and groundwater transport, and it treats all major exposure pathways. MEPAS also incorporates some of the features of the EPA models for particulate suspension by mechanical and wind-driven erosion. However, there is not an intrinsic Monte Carlo capability for uncertainty analysis.

Because Battelle Memorial Institute declined our request for permission to examine portions of the MEPAS source code, however, we were not able to consider the MEPAS program at this time for application to the Rocky Flats site soil contamination

GENII

The GENII code was designed by Pacific Northwest Laboratory to address exposure and dose resulting from both routine and accidental releases of radionuclides. Doses can be calculated on an annual, committed, or accumulated basis. GENII models the same pathways that are included in the RESRAD simulations that were used in the previous soil action levels document. These pathways are resuspension and inhalation of contaminated soil, inadvertent soil ingestion, transfer of radioactivity into homegrown produce and animal products, and external exposure of the subject to surface soil contamination and contaminated airborne particles. Two resuspension models are available in GENII: a mass loading approach that is similar to the one in RESRAD

GENII also has available a scenario of someone offsite who has been exposed to radioactivity that has been released and transported from a remote location. The RESRAD code

148

The main focus of the report was the evaluation of computer programs for use in the project. The models reviewed in this report are RESRAD, MEPAS, GENII, MMSOILS, and DandD. The Department of Energy (DOE) calculated soil action levels with the RESRAD program (Version 5.61) previously, and part of the scope of this current project is to review their calculations for choice of the parameter values used in RESRAD. *RAC* selected programs that were generally comparable to RESRAD and that are widely used. All five programs have been developed under sponsorship of one or more federal agencies.

We selected the programs using these criteria:

(1) Correctness of the mathematical models: that is, how well does the model account for exposure pathways and site features, and how consistent is the program with site-specific data.
(2) Validation of the programs: that is, has the program been checked or confirmed with data that is well documented.
(3) Source Code: that is, how available is the entire computer code to *RAC* and has the program been documented.
(4) Platform (i.e., computer and operating system) and programming language.
(5) Flexibility of operating features, that is, what is the possibility of bypassing the automatic user interface in order to specify input and output files.

Another consideration in selecting computer programs for the study was our desire to use state-of-the-art methods for carrying out work, especially by incorporating uncertainty estimates in our work. We have developed a method to calculate soil action levels that incorporates uncertainty into the process. The term uncertainty usually implies lack of knowledge about the value of a model parameter or the accuracy of a model prediction. We represent these uncertainties as probability distributions. Because inputs to the selected code will be in the form of probability distributions, we have carefully considered how suitable the various computer programs will be at providing a distribution of results for dose, or soil action levels.

All five of the programs selected for evaluation can be installed and executed under some version of the Microsoft Windows operating system and, as a result, all of the programs are accessible. RESRAD was developed by DOE to evaluate the clean up and remediation of radionuclide-contaminated soils at DOE facilities. MEPAS, which was developed at Pacific Northwest Laboratories (PNL) and is now commercially marketed, is applicable to radioactive and nonradioactive pollutants in many environmental media. GENII, also developed at PNL, provides internal and external dose estimates for exposure through all pathways that are ordinarily considered in environmental radiological assessments. GENII has been under development for more than a decade and is unlikely to be modified further by its developers. MMSOILS, which was developed for the Environmental Protection Agency, is a large multimedia environmental transport program that was designed for screening assessments of chemical contamination. Although it does not treat radioactivity and decay chains, it was included in this review because it could possibly be useful for radionuclides in soils. DandD is currently under development by the NRC as a screening level code for decontamination and decommissioning of NRC-regulated facilities. Each of the programs are described briefly to show how they might be used or not in the current project.

149 **DRAFT**

REVIEW OF THE RADIONUCLIDE SOIL ACTION LEVELS AT THE ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE

TASK 2: COMPUTER MODELS

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 about 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. This current project is evaluating the radionuclide soil action levels developed for implementation by the DOE, the Environmental Protection Agency (EPA) and the Colorado Department of Public Health and Environment (CDPHE). A soil action level is a concentration of a radionuclide in the soil established to protect people from receiving radiation doses above a set limit. As a result of public concern about the proposed soil action levels, DOE provided funds for the Radionuclide Soil Action Level Oversight Panel (RSALOP) to select a contractor to conduct an independent assessment and to calculate soil actions levels for the RFETS. *Risk Assessment Corporation* (*RAC*) was selected to carry out the study.

The goal of the Task 2 report is to discuss and compare environmental assessment programs that might be used for developing soil action levels for the Rocky Flats Environmental Technology Site (RFETS). In addition, the Task 2 report discusses other important aspects involved in calculating soil action levels. The soil action levels depend on four things:

(1) How radioactive material is transported in the environment to people (transport pathways)
(2) How people might be exposed to the radioactive materials (exposure scenarios)
(3) How radiation dose to a person is assessed (radiation dosimetry)
(4) How radiation protection guidelines fit in (annual dose limits).

Because of these considerations, the report explains the importance of creating valid exposure scenarios for the project, and discusses several factors that are important in the transport of radioactive materials in air and water in an area like Rocky Flats. In designing the scenarios, we carefully considered offsite exposures so that if the person living onsite full-time is protected, then the person living offsite will be protected. Understanding the behavior of radionuclides in the soil and how soil can be disturbed or resuspended is an integral part of the project since inhalation is the major exposure pathway for this work. Nevertheless, the potential significance of the groundwater pathway has been carefully evaluated. The discussion in the Task 2 report shows that groundwater is an extremely complex pathway. *RAC* will not assess it in significant detail in the soil action level project because of the extensive ongoing Actinide Migration research. We are including groundwater as a pathway in one of our scenarios to provide a bounding level, screening calculation with contaminated drinking water as a pathway for dose. Some of the topics touched upon in the Task 2 report are fully explained in subsequent reports for the project.

memo

Date: 7/22/99

To: Risk Assessment Corporation

From: Carla Sanda

RE: COMMENTS TO TASK 2: COMPUTER MODELS - GENERAL SUMMARY

The General Summary seems to capture the primary points of the report and takes a clear, less-technical approach. I believe that the General Summary portions of each of the Task Reports could serve as project updates for distribution at the public meetings, with just a few minor changes in format. Therefore, I recommend that each of the General Summaries be formatted with text falling into the following sections: Background, Goals/Objectives, Results/Accomplishments (or Findings), Conclusions.

I have only a few minor changes to this specific summary, as follows:

- Consider writing the summary in the third person; i.e., rather than "we carefully considered", perhaps "*RAC* carefully considered".
- Is there a definition for the acronyms GENII and MM Soils that could be provided?
- Page 2 – paragraph beginning with "All five of the programs selected for evaluation.....": Delete everything from the sentence betinning with "RESRAD was developed by DOE...." through to, and including, the sentence beginning "Although it does not treat radioactivity and decay chains....". This seems to be redundant information, in that the description for each of the programs that follows provides similar information.

151

ADVANCED INTEGRATED MANAGEMENT SERVICES, INC.

AIMSI

FACSIMILE TRANSMITTAL SHEET

TO: RAC -- JOHN, KATHLEEN, JILL	FROM: FROM: CARLA SAND
COMPANY:	DATE:
FAX NUMBER:	TOTAL NO. OF PAGES INCLUDING COVER: 6
SENDER'S PHONE NUMBER: 303-456-0884	SENDER'S FAX NUMBER: 303-456-0858
RE:	

☐ URGENT ☑ FOR REVIEW ☐ PLEASE COMMENT ☐ PLEASE REPLY ☐ PLEASE SIGN

NOTES/COMMENTS:

Enclosed are the matl's from LeRoy Moore - Re: Comments to Task 3.

Carla

5460 WARD ROAD, SUITE 370, ARVADA, COLORADO 80002
PHONE: 303-456-0884
FAX: 303-456-0858

152

To: CarlaSanda
From: leroymoore@earthlink.net (LeRoy Moore)
Subject: Comments on TASK 3 Draft Report on Inputs and Assumptions

My comments are given page by page.

p. v, Ex. Sum: about three-fourths down in the opening paragraph a sentence begins: "As a result of public concern about the proposed soil actions levels. . . ." Delete "proposed" and change to read: "soil action levels adopted in October 1996."

p. 1: Change opening sentence of Introd to read: "Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which remedial action would be required to prevent people from receiving doses above an officially designated level."

pp. 1-3: Why is RAC using RESRAD 5.82 rather than 5.61? My recollection is that at one meeting a couple of months ago RAC presented us with the disturbing info that 5.82's parameters had been so modified that feeding in the same data used by the agencies in setting the original RF RSALs resulted in much higher allowed concentrations of Pu, etc. The text on pp. 2-3 (esp. Table 1 on p. 3) repeats this info. We go from a RSAL for Pu of 1429 pCi/g to one of 8351, which, to put it mildly, is outrageous. I do not recall that the Panel asked RAC to proceed with 5.82. I do recall that there was a request for documentation from DOE of the instructions they gave to Argonne along with their request that RESRAD be updated. Have we received this documentation? Short of getting it and thus understanding why the outcome from calculations is so much higher on the revised RESRAD, I think we should stay with the program used by the agencies initially. Is there any reason we cannot do this?

p. 2, second para. under "Difference between versions": Why use a value for annual mean for Denver area wind speed derived from a National Climatic Data Center report? Isn't there site-specific data for wind speed at RF? RAC may recall that wind is stronger at RF than in Denver, and that the prevailing wind blows in a different direction. The RF original siting resulted from a mistake about wind, namely, that it was based on readings done in Denver, not at RF itself.

p. 8: Contrary to what is said in the first full paragraph, Litaor thought he found Pu in particle and colloidal form moving with groundwater in May/June 1995. He at least speculates, as I understand his work, that anoxic conditions of soil saturation may release some Pu into dissolved form. The second full para. on this page refers to this aspect of Litaor's work, but I wonder if it's correct to suggest that subsurface storm flow could be important only for "localized soil contamination areas," since seeps release material into stream channels that go to holding ponds or eventually exit the site. Also, it's not clear that channels have been adequately analyzed in terms of their ability to hold material flowing throught them; that is, do they leak?

p. 30: My note above about wind may be answered from RAC's perspective on pp. 29-30. But I raise a further question regarding RAC's assumption that "high winds will not be explored further in the SAL project." Why? Evidently because wind blows contamination away and thus lessens possibility of future resuspension by this means. OK. This makes sense, though it's not very reassuring news. But a decision to set aside further analysis re. wind seems predicated on the assumption that the 903 Pad will not release more and that main sources of resuspension have been already depleted. What about remediation of 903 area? What about taking down of buildings and exposure of whole new areas of contaminated soil? What about any construction activities that may occur? There seems to be ample reason to keep airborne resuspensioon alive as a very likely pathway for future exposure of unwitting populations. Am I missing something here?

153

p. 31: Re. scenarios, one peer reviewer in commenting on Task 2 raised a serious question re. "institutional controls." In a May 7, 1999, memo to RAC I raised the issue as follows: "One of the peer reviewers for the independent assessment of the Rocky Flats RSALs states that the RSALs as adopted misapply the concept of 'institutional controls' in relation to the 15/85 mrem/year dose (see attached 'Review Comments on the March 1999 Draft Report . . . for Task 2: Computer Models," section 1, 'Application of the 85 mrem/y criterion'). This suggests that the Rocky Flats RSALs violate CERCLA in the way the 'institutional controls' concept is employed. What corrections need to be made?" I raise this question anew because it was not previously answered and because it comes up again under "scenarios." One of the scenarios included in the officially adopted RSALs -- the hypothetical future resident -- assumes disappearance of institutional controls, in possible violation of CERCLA, if the peer reviewer is correct in the comment submitted. If the reviewer is correct, then the hypothetical future resident scenario (as well as all other hypothetical future scenarios) needs to be recast in terms not of a possible dose of 85 mrem/yr but of 15. How does RAC respond?

pp. 34-36: This section does not make sense to me. Table 11 shows breathing rates ranging from 7.5 L/min to 712. Is this correct? The numbers given on p. 36 seem far less than those provided by Joe Goldfield January 31, 1999, paper. Joe's paper has the virtue of clarity and persuasivness. I defer to him in the hopes he will make a clear response to this section.

pp. 37-40: Re. soil ingestion. I again defer to Joe Goldfield.

154

To: RAC
From: LeRoy Moore
Date: May 7, 1999
Re: A question that emerges from comments of a peer reviewer on Task 2, Computer Models

One of the peer reviewers for the independent assessment of the Rocky Flats RSALs states that the RSALs as adopted misapply the concept of "institutional controls" in relation to the 15/85 mrem/year dose (see attached "Review Comments on the March 1999 Draft Report . . . for Task 2: Computer Models," section 1, "Application of the 85 mrem/y criterion"). This suggests that the Rocky Flats RSALs violate CERCLA in the way the "institutional controls" concept is employed. What corrections need to be made?

cc: Rocky Flats RSAL Oversight Panel

155

Review Comments on the March 1999 Draft Report by the Risk Assessment Corporation (RAS) for Task 2: Computer Models

This is a carefully prepared and mostly excellent draft. Prior to commenting on the Task 2 Report, this reviewer reviewed several background documents: the DOE report "Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement - Final, Oct. 31, 1996, and its accompanying "Responsiveness Summary;" the RAS draft report for Task 1, Feb. 1999; the report by Joseph Goldfield entitled "Breathing Rates of Exposed Persons Residing on Plutonium Contaminated Soil for Calculating Health Effects;" and two papers by LeRoy Moore entitled "Action Levels for Radionuclides in Soils for Cleanup of Rocky Flats" and "Seven Reasons for an Independent Review of the Rocky Flats Action Levels." Review of the first of these reports raised a number of concerns regarding the assumptions underlying their application of the 15/85 mrem/y dose criteria and their choice of exposure scenarios for implementing those criteria via soil action levels, including the selection of parameters characterizing the environment and individuals exposed. I was pleased to find that the authors of the Task 2 draft report reflected many of the same concerns.

By way of background for comments on the Task 2 report, the following summarizes my concerns with the DOE report:

1. <u>Application of the 85 mrem/y criterion</u>.

There is a conspicuous absence of a clear statement of the limited use of the 85 mrem/y criterion intended by EPA, and a strong implication that it is being misused. This criterion was proposed by EPA as an 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. 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 land uses in the distant future for situations in which institutional controls are *assumed* to no longer exist. To the contrary, CERCLA regulations require the lead agency to review the efficacy of institutional controls *no less often than every five years* for as long as they are required to maintain conformance with the level permitting unrestricted use (in this case 15 mrem/y)(see 40 CFR Part 300.430(f)(4)(ii)). 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 under the above periodic review requirement.

It is not obvious to this reviewer, especially for the two types of buffer areas (these are not differentiated in the DOE report), but also for the industrial area, that either the commitments or assurances of effectiveness for the necessary institutional controls exist. The DOE report depends on the documents "Action Levels and Standards Framework for Surface Water, Ground Water, and Soils" (ALF) and the "Rocky Flats Vision." These documents, as well as the "Rocky Flats Cleanup Agreement" (RFCA) and proposed "Modifications to the Action Levels

156

and Standards Framework" were not available for this review. However, a "vision" is not a legal commitment, and the discussion of near and intermediate term land uses and, more significantly, the absence of any discussion of long-term land use in the last paragraph on p. 6-15 of the DOE report creates the impression that the state of commitments for and assurances of effectiveness of institutional controls in the future is very uncertain.

The implication of the above, given the long-term contamination present at Rocky Flats, is clear. If the lead agency (DOE), State, and local officials cannot commit to and provide reasonable assurance of maintaining necessary institutional controls in an effective manner for 1000 years, then consideration must be given to cleanup of the site now to levels that would meet 15 mrem/y in the absence of such controls. Obviously, this point is critical to choosing the Tier I Action Levels for the so-called "buffer" and "industrial" areas.

There is also a need to develop a Tier I level applicable *outside* the buffer areas, since these locations must meet the 15 mrem/y criterion under unrestricted use (presumably under a rural or rancher residential scenario), and the action levels for the immediately adjacent buffer area, at least under the current proposal, would permit significantly higher levels. As noted above, if the necessary assurances for long-term institutional control cannot be met for the buffer and/or industrial areas, this level should apply there also.

2. Exposure Scenarios:

Under CERCLA, the choice of exposure scenarios is intended to assure protection of the "*Reasonably Maximum Exposed*" (RME) individual. This is not the same as the *average* member of the affected population, nor is it the *most* exposed individual. EPA has devoted considerable effort to clarifying this admittedly elusive concept. The following quotes are typical of EPA 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... The intent of the RME is to estimate a conservative exposure case (i.e., well above the average) that is still within the range of possible exposures.*" ("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 or estimated) distribution. The conceptual range is not meant to precisely define the limits of this descriptor, but should be used by the assessor as a target range for characterizing "high-end" risk.*" ("Guidance on Risk Characterization for Risk Managers and Risk Assessors," Memo from F. Henry Habicht II, Deputy Administrator, EPA, to Assistant Administrators and Regional Administrators, February 26, 1992.

157

DRAFT FINAL REPORT

Task 2: Computer Models

Radionuclide Soil Action Level Oversight Panel

July 1999

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

"Setting the standard in environmental health"

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158

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159

CONTENTS

FIGURES

161

TABLES

162

REVIEW OF THE RADIONUCLIDE SOIL ACTION LEVELS AT THE ROCKY FLATS ENVIRONMENTAL TECHNOLOGY SITE

TASK 2. COMPUTER MODELS

Abstract

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

1. INTRODUCTION

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

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

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

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

164

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

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

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

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

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

\165

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

2. SOIL ACTION LEVELS

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

The soil action levels depend on four things:

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

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

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

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

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

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

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

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

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

168

2.1 Formulation

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

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

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

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

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

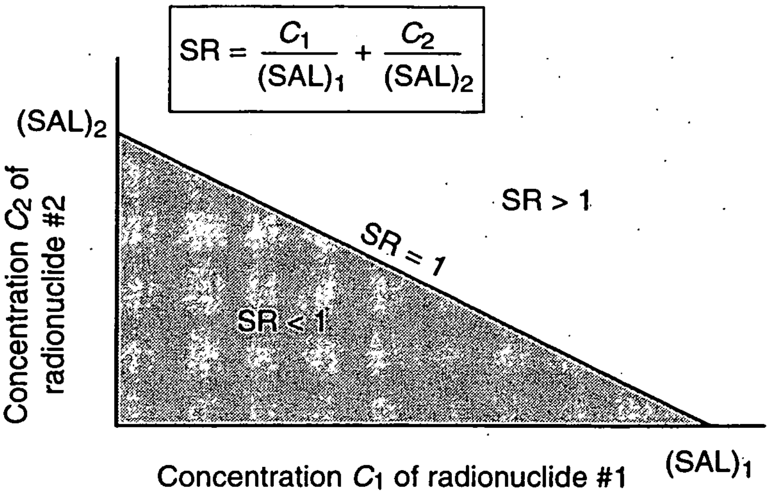

Figure 2.1-1. Geometric interpretation of the sum of ratios (SR) for two radionuclides (N = 2) and one scenario (S = 1). All points (C_1, C_2) on the line represent pairs of concentrations for which the sum of ratios equals 1. For all points in the shaded rectangle beneath the line, the pair of concentrations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

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

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

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

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

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

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

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

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

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

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

170

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

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

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

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

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

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

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

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

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

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

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

and the sum of ratios for scenario s as

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

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

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

2.2 Stochastic SALs

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

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

172

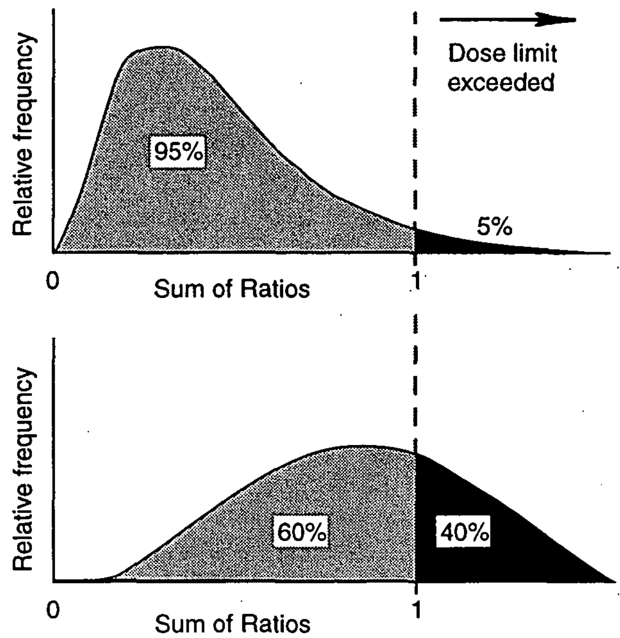

Figure 2.2-1. Schematic illustration of uncertainty distributions for the sum of ratios of soil concentrations divided by the corresponding soil action levels. In the top panel, the probability is 5% that the dose limit for a scenario would be exceeded. In the bottom, the probability is 40%.

2.3 Exposure scenarios

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

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

For the RSAL assessment, some of the parameters are breathing rates for various activity levels and ages, soil ingestion rates for children and adults, fraction of time spent indoors and outdoors, and the potential use of or exposure to contaminated water from the area. Selecting appropriate parameters for the scenarios depends upon a thorough review of the scientific literature and fully considering the uncertainty (or variability) distributions of the relevant parameters. We use a wide range of references and studies to compile information on parameters. Subsequently, we can generate a distribution of values and sample from the distribution, using Monte Carlo techniques. This process considers the available studies equally. The distributions can be characterized with a central value such as the median and some measure of the spread of the distribution, such as the standard deviation or the 5th and 95th percentiles of the distribution. In developing a particular scenario and considering variability of a parameter within the population studied, we can use a high (or low) percentile of the distribution as needed to extend protection to a larger fraction of a potentially exposed population with characteristics similar to those of the scenario subject. Once a parameter value is selected from our distribution of values for use in the scenario, the scenarios are considered fixed just as standards are fixed as a benchmark against which to measure an uncertain value. Behavioral characteristics should be plausible and relevant to the exposure situations and the radiation protection objectives.

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

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

174

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

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

1. The future residential exposure scenario assumes that an individual resides onsite all year and grows and consumes homegrown produce. This person would be exposed to radioactive materials in soils by directly ingesting the soils, by inhaling resuspended soils, by external gamma exposure from contaminated soil and airborne radioactivity, and by ingesting produce grown in contaminated soil. This scenario is from the current Rocky Flats Cleanup Agreement.
2. The open space exposure scenario assumes the person visits the site 25 times per year for recreational purposes, spending 5 hours per visit at the site. The person would be exposed to radioactive materials in the soil by directly ingesting the soils, by inhalation of resuspended soils, and by external gamma exposure from the soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement.
3. The office worker exposure scenario represents an individual who works a 40-hour per week, 50-week per year job indoors in a building complex at the site. It is assumed that this person would be exposed to radioactive material in soils by directly ingesting the soils, by inhaling resuspended soils, and by external gamma exposure from soils and airborne radioactivity. This scenario is from the current Rocky Flats Cleanup Agreement.
4. The resident rancher scenario assumes future loss of institutional control. The rancher is raising a family, maintaining a garden and leading an active life at the site, spending 24 hours per day, 365 days per year or 8760 hours at the site. Of that time, over 40% is spent out of doors. The potential pathways of exposure for this person include inhalation; eating produce from garden irrigated with groundwater, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils and airborne radioactivity. The annual breathing rate is 10,800 m^3 per year, based on a time-weighted average of breathing rates and activity levels as described during the monthly RSALs meetings. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
5. Infant in rancher family is 0 to 2 years of age, and onsite 24 hours per day, 365 days per year, or 8760 hr/year. The infant's potential pathways of exposure include inhalation, some direct soil ingestion from outdoor activities, and direct gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
6. The child of the rancher family is assumed to be 5 to 17 years of age, and onsite 24 hours per day, 365 days per year, or 8760 hr/year. The potential pathways of exposure include inhalation, eating produce from garden irrigated with water from a stream on the site, direct soil ingestion, and gamma exposure from soils and airborne radioactivity. *RAC* proposed this scenario for consideration at the January 1999 RSAL meeting.
7. The current onsite industrial worker scenario assumes a person works onsite 8½ hours per day, 5 days per week, 50 weeks a year, or 2100 hours per year. It is assumed that 60% of the worker's time is spent outdoors. The potential pathways of exposure for this person include inhalation, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils. The annual breathing rate is 3700 m^3 per year, based on a time-weighted average of breathing rates and activity levels for the time spent onsite. *RAC* proposed this scenario for consideration at the February 1999 RSAL meeting.

176

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

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

[a] NA = not applicable.

177

3. SITE CONCEPTUAL MODEL

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

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

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

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

3.1 Transport pathways

3.1.1 Availability of residual radioactivity in surface soil over time

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

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

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

179

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

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

3.1.2 Spatial disaggregation of soil

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

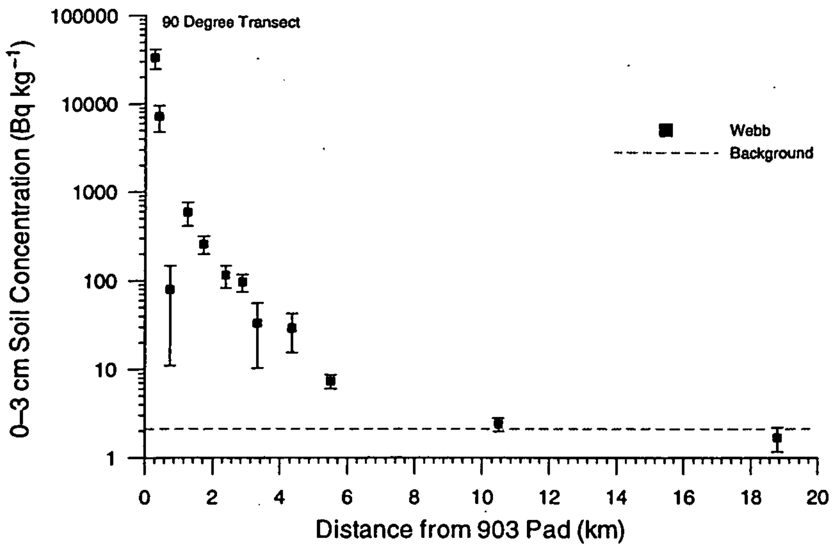

Figure 3.1.2-1. Plutonium-239 concentrations in soil (Bq kg^{-1}) at RFETS along a 90° transect (eastward) from the 903 Pad area. The data are from Webb (1996).

3.1.3 Resuspended contaminated soil

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

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

189

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

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

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

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

182

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

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

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

3.1.4 Groundwater and surface water transport

In calculating the proposed soil action levels (DOE/EPA/CDPHE 1996), the groundwater and surface water pathways were dismissed because (1) surface water features (Woman and Walnut Creeks) on the site are perennial and would not provide a reliable year-round water source for an individual living on the site and (2) surface aquifers underlying the site do not produce enough

183

water for domestic or agricultural use. In addition, the aquatic food pathway was eliminated because the streams are not capable of sustaining a viable fish population. In this section, we will discuss these assumptions and the rationale behind them, and we will examine the ramifications of dismissing the groundwater and surface water pathways in the assessment.

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

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

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

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

The LAHU is composed of fine to medium grained sandstone separated by a few claystone beds in the upper portion. Thickness ranges from 200 to 220 feet for the "A" and "B" sandstone that comprise the lower interval of the Laramie formation, and 80 feet for underlying Fox Hills sandstone unit. The Laramie–Fox Hills aquifer system is the target of most water wells in the vicinity of Rocky Flats because this aquifer provides sufficient water for domestic and industrial uses. Recharge to the aquifer takes place along the foothills west of the RFETS where the permeable sandstone beds of the formation are folded up and exposed. The permeable sandstone generally dips eastward toward the center of the Denver Basin.

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

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

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

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

185

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

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

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

a. Source: DOE 1996a

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

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

3.2 Exposure Modes

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

186

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

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

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

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

187

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

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

4. CANDIDATE COMPUTER PROGRAMS

4.1 Introduction

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

(1) Presumed correctness of the models implemented by the programs, as indicated by their general acceptance, logical correspondence with features of the site, treatment of exposure pathways, and consistency with the available site data
(2) Amount and quality of validation that has been carried out and documented, and suitability for validation with local data
(3) Quality of program documentation and availability of source code
(4) Platform (i.e., computer and operating system) and (if source code is made available) programming language
(5) Flexibility of operating features, particularly the possibility of bypassing the user interface in order to invoke the computational part of the program and specify input and output files from the command line.

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

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

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

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

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

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

4.1.1 Verification of Computer Programs

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

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

140

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

4.1.2 Validation of Computer Programs

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

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

The interpretation of validation exercises is never entirely clean. Consider once again the example of predicting uranium concentrations in air. Our calculations involve more than the computer program: there are the estimates of the uranium releases, which are subject to error, and there are meteorological data, which may or may not be accurate for the locations and period for which they were applied. It is possible for errors in the data to compensate for errors in the model, giving apparently good results and encouraging us to trust a program that intrinsically might not be an acceptable representation of the processes we are simulating. Alternatively, errors in the data could make an acceptable model look bad. When we must depend on data that are available, it is practically impossible to implement rigorous designs that might remove these confounding effects. We must generally be satisfied with making as many tests of two or more correlated functionalities (e.g., diffusion and deposition, if we have data for both) as possible, in the hope that good agreement of predictions and data will be persuasive at an admittedly subjective level.

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

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

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

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

192

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

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

4.2 RESRAD

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

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

4.2.1 RESRAD overview

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

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

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

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

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

194

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

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

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

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

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

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

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

4.2.2 Code acquisition

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

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

196

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

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

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

4.2.3. Changes in the area factor for resuspension

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

Discussion of these changes and the related mechanisms is of necessity somewhat technical. The changes involve the calculation of the area factor, which affects resuspension predictions. The area factor accounts for the dilution of locally contaminated airborne dust by uncontaminated dust resuspended from outside the contaminated area. Larger (smaller) area factors correspond to larger (smaller) predictions of airborne contamination, which would produce larger (smaller) predictions of dose by inhalation and by external exposure to airborne gamma-emitting radionuclides. Bearing these relationships in mind, some readers may prefer to refer primarily to Figure 4.2.3-1 for a sense of the extent to which the changes might reduce RESRAD predictions of air concentration.

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

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

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

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

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

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

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

198

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

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

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

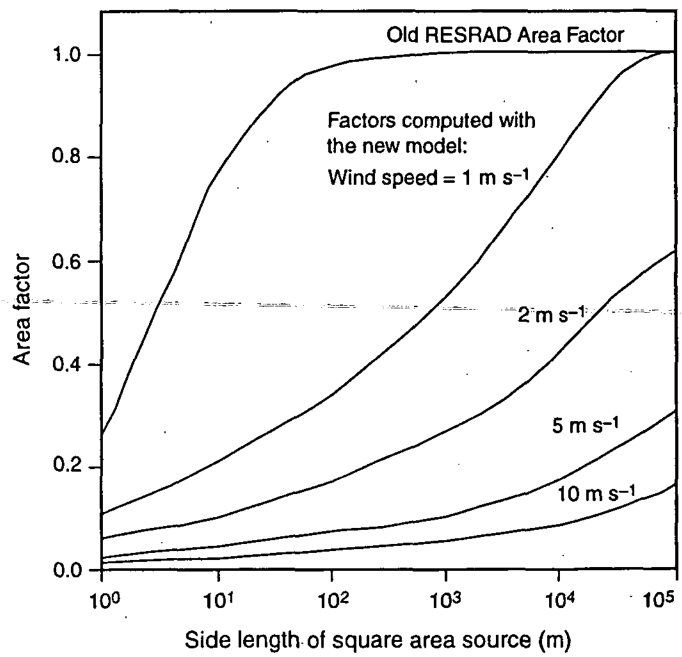

Figure 4.2.3-1. Comparison of the old and new RESRAD area factors for particle size 1 μm, plotted against the side length of a square contaminated area. The new area factor is shown for several values of the wind speed. This figure was redrawn from Chang et al. (1998).

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

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

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

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

200

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

4.3 MEPAS

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

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

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

4.4 GENII

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

4.4.1 Code overview

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

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

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

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

202

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

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

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

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

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

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

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

203

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

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

4.4.3 Code acquisition and testing

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

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

204

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

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

a. from DOE (1996), Attachment I

b. Area of contamination in GENII is only defined in terms of less than or greater than 1250 m^2

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

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

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

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

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

a. [(DSR (RESRAD) − DSR (GENII)]/DSR (RESRAD)

4.5 MMSOILS

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

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

206

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

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

4.6 DandD

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

4.6.1. Code overview

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

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

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

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

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

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

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

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

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

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

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

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

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

208

4.6.3. Code acquisition and testing

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

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

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

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

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

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

[a]DandD input units shown; this converts to the same value as the RESRAD parameter.

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

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

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

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

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

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

210

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

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

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

5. CONCLUSIONS AND RECOMMENDATIONS

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

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

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

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

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

We have some recommendations for DOE and the developers of RESRAD. We are aware that the evolving Windows graphic user interface (GUI) is intended to make the program more accessible to a variety of users, but this greater utility comes at a cost to some potential users. It often is desirable to link programs together, with outputs from one becoming inputs to another. The procedure is usually implemented by writing scripts, which are control programs for the process (Unix operating systems are particularly hospitable to this approach). But a GUI defeats script-driven executions. We are not suggesting that the GUI be eliminated, because it is probably the preferred access for the majority of users, but we do urge DOE and the RESRAD developers to facilitate a way of bypassing the GUI and launching RESRAD from the command line.

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

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

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

223

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215

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DRAFT REPORT

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

July 1999

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

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217

DRAFT REPORT

TASK 3: INPUTS AND ASSUMPTIONS

Radionuclide Soil Action Level Oversight Panel

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errata . . . .

Page 13, paragraph 1, sentence 2 (line 3) reads: "We will, however, provide a bounding level, screnning calculation for a single scenario (DOE/EPA/CDPHE resident)...", but should read: "We will....for a single scenario (RAC resident rancher)..."

218

RAC Report No. 15-RSALOP-1999-DRAFT

DRAFT REPORT

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

July 1999

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

219

CONTENTS

FIGURES

1221 **DRAFT**

TABLES

DRAFT

EXECUTIVE SUMMARY

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

RAC is using several environmental assessment computer programs, in particular, the RESRAD computer program, to calculate the soil action levels for this project. The purpose of Task 3, *Inputs and Assumptions*, is to evaluate the input parameters and assumptions for their importance in determining the dose and soil action levels for cleanup at the RFETS. The task involves performing a sensitivity analysis using RESRAD to identify those parameters that have the greatest impact on the outcome of the soil action level calculation. For the parameters that are the most important to the final outcome, the task requires that *RAC* develop site-specific values if data are available or to create uncertainty distributions of values from published literature sources. The sensitivity analysis was a single-parameter analysis, where a range of values for one parameter at a time was evaluated. *RAC* used the latest version of the RESRAD code (Version 5.82) to carry out the sensitivity analysis. This version is an update from the version used in the previous soil action level assessment (DOE/EPA/CDPHE 1996).

Of over 50 parameters assessed for their influence on the final result, five parameters were found to impact the final result to the greatest extent. These parameters are:

- distribution coefficient
- soil-to-plant transfer factor
- area of contamination
- mass loading factor
- mean annual wind speed.

Most Sensitive Parameters

The majority of the report focuses on these five parameters and provides parameter values or uncertainty distributions for them based on site-specific data or on literature values. The uncertainty distributions describe the variability in the values that occurs from natural variability or from lack of knowledge about a particular parameter. The following table summarizes the differences in parameter values or approach between the previous DOE/EPA/CDPHE assessment and the RAC approach.

Table ES-1. Values for the Five Most Sensitive Parameters for the Independent Calculation and Comparison with those from Previous Assessment

Parameter	*RAC* value	DOE/EPA/CDPHE value
Distribution coefficient	Treated stochastically based on Rocky Flats measurements; median values (GSD) of	Deterministic
	Pu = 218 cm^3 g^{-1} (1.16)	Pu = 218 cm^3 g^{-1}
	Am = 76 cm^3 g^{-1} (2.52)	Am = 76 cm^3 g^{-1}
	U = 218 cm^3 g^{-1} (3.92)	U = 50 cm^3 g^{-1}
Soil-to-plant transfer factors	Treated stochastically based on NCRP Report 129 data; median values (GSD)	Deterministic
	Pu = 1.0 x 10^{-3} (2.5)	Pu = 1.0 x 10^{-3}
	Am = 1.0 x 10^{-3} (2.5)	Am = 1.0 x 10^{-3}
	U = 2.0 x 10^{-3} (2.5)	U = 2.0 x 10^{-3}
Area of Contaminated Zone	Defined based on historic soil concentration measurements at Rocky Flats	40,000 m^2
Mass loading	Model will be calibrated based on results of soil and airborne concentration	0.000026 g m^{-3}
Mean annual wind speed	Treated stochastically based on annual average wind data	Not required for RESRAD V 5.61

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

The distribution coefficient is important in the Radionuclide Soil Action Level (RSAL) assessment because it defines the relationship of the concentration of the contaminant in the soil to the concentration of the contaminant in water, and can influence calculations involving contaminants in the groundwater. *RAC* included groundwater as a source of water in the rancher scenario so it is important to carefully consider all data in establishing a value or range of values for this parameter. The distribution coefficient, called the K_d value, can extend over a very wide range even for a single type of soil so it is important to incorporate as much data as possible in our assessment. We have expanded the distribution coefficients reported previously by creating a distribution of values for uranium, plutonium, and americium, based on a further review of the literature. The midpoint of our uncertainty distributions for the radionuclides is the midpoint used in the previous assessment, except for uranium where our midpoint, or geometric mean, is four times higher. In our assessment, the distribution for each radionuclide is further defined by the geometric standard deviation, which gives an estimate of how much uncertainty there is about the midpoint.

Soil-to-plant transfer factors quantify that portion of contamination in soil that is transferred to plants via root uptake. The previous DOE/EPA/CDPHE assessment used a deterministic approach, while *RAC* treats these factors stochastically based on the recent National Council on Radiation Protection and Measurement Report, *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies* (NCRP 1999). This screening methodology suggests distributions for soil-to-plant transfer factor that reflect uncertainty resulting from different soil conditions, soil types, and soil chemistry.

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

Mass loading is a measure of resuspension of soil from the ground. It is a complex process that is affected by many environmental factors that have not been well quantified The previous DOE/EPA/CDPHE assessment used a value of 0.000026 g m^{-3} for mass loading factor as a representation of resuspension. The current version of RESRAD uses a mass loading factor to define resuspension but even the developers of RESRAD stressed its inadequacy at representing actual conditions at a given site. As a result, *RAC* is using historic air monitoring data as the best measure of resuspension. *RAC* will consider the location of each scenario onsite where the hypothetical person resides and/or works, and use actual air monitoring data in combination with the soil contamination data described above to set up a relationship between concentrations in air and soil that can be used to estimate resuspension. This process bypasses the area factor calculation in RESRAD and defines resuspension based on actual air monitoring data.

The mean annual wind speed was not required in the previous version of RESRAD so the previous assessment does not specify a value for this parameter. However, the wind speed is important in estimating resuspension in the current RESRAD version. Because RAC is estimating resuspension based on site-specific air monitoring data, it is important to use site-specific meteorological data, too. *RAC* is using a 5-year average wind speed and atmospheric stability class information from the onsite Rocky Flats meteorological station. High wind events occur in the Rocky Flats area and were evaluated in the Historical Public Exposure Studies on Rocky Flats for their effect on moving contamination from the 903 Area before it was covered with an asphalt pad. High wind also results in lower air concentration than would be expected if the same material was dispersed over a longer period of time during average wind speed conditions. As a result, high wind events are not evaluated separately in this assessment. Rather, a distribution of wind speed values will be used based on measurements at the Rocky Flats weather station.

Less Sensitive Parameters

Five parameters were found to affect the outcome of the calculation only slightly. These parameters are the cover depth (depth of soil that must be removed to reveal the contaminated soil), the fraction of the total outside air contamination that is available indoors (indoor dust filtration), the depth of surface soil available for resuspension, the fraction of irrigation water contaminated by groundwater, and the thickness of the contamination in the soil. For these somewhat sensitive parameters we used the values from the current DOE/EPA/CDPHE assessment for cover depth and indoor dust filtration. For the other three, we selected a value more consistent with studies published in the open scientific literature. For the depth of surface soil available for resuspension (depth of soil mixing layer), we selected a value of 0.03 m, instead of 0.15 m, based on published studies that define the surface of resuspendable soils. For the thickness of the contaminated zone, we selected a value of 0.20 m, instead of 0.15 m, based on studies that show the contamination is distributed over the top 20 cm (0.20 m) of soil with very little movement of the contamination over the past 20 years. For the fraction of irrigation water contaminated by groundwater (irrigation water contamination fraction) we determined that groundwater might be used for irrigation purposes or as a source of drinking water. As a result, we assumed that all of the groundwater used for irrigation would be contaminated (irrigation contamination fraction = 1.0). In the previous assessment, it was assumed that none of the water would be contaminated (irrigation contamination fraction = 0).

Other Parameters

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

The report also summarizes current studies that clearly show that plutonium in the soil at Rocky Flats is insoluble and thus may not get into the groundwater. However, *RAC* has included the groundwater pathway in the rancher scenario, and this report describes our approach to studying the sensitivity of the drinking water pathway when contaminated groundwater is assumed as the source. This assessment shows that groundwater can have an impact on dose that needs to be recognized. Because of the severe limitations on time and resources in this study, we can only recommend that a future study be directed toward this type of work, particularly looking at the migration of ^{241}Am and its daughters.

Another important parameter for RESRAD is the initial concentrations of radionuclides. In the previous assessment, DOE/EPA/CDPHE defined the initial concentrations of each radionuclide of interest as 100 pCi per gram. In contrast, *RAC* uses the available published literature in combination with measured soil concentration data to determine actual soil concentrations, initialized to the year that the soil action level calculations begin. The concentrations of ^{238}Pu, ^{239}Pu, ^{240}Pu, ^{241}Pu, ^{242}Pu, ^{241}Am, ^{234}U, ^{235}U, and ^{238}U are given relative to ^{239}Pu. *RAC* uses soil concentration data to determine current values for ^{239}Pu. This technique clarifies the RESRAD results for the user by making the calculation of dose more meaningful. The report also provides the most recent values for inhalation and ingestion dose conversion factors that will be used in the independent calculation in Task 5.

Scenarios

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

Some scenario related parameter values are different from those in the previous assessment. Because we include the drinking water pathway in our assessment, we provide an annual drinking water intake of 730 liters per year. The current DOE/EPA/CDPHE scenarios do not include drinking water as a potential pathway. We recommend higher annual consumption rates for fruits, vegetables, and grains based on published literature values than those used in the previous assessment.

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

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TASK 3: INPUTS AND ASSUMPTIONS

INTRODUCTION

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

The primary goal of Task 3 is to report the results of a sensitivity analysis conducted on the inputs and assumptions required for the use of RESRAD. Site-specific values were derived or uncertainty distributions were created for critical parameters emerging from the sensitivity analysis. The sensitivity of each parameter was assessed using the built-in Monte Carlo-based sensitivity analysis packaged with the latest version of RESRAD. Also included in the Task 3 report is the careful evaluation of scenarios for their applicability to potential future land uses. This report describes the process of scenario evaluation and reports the scenarios chosen for the independent analysis.

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

Parameters Explored

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

- Mass loading factor
- Area of contamination
- Mean annual wind speed
- Soil-to-plant transfer factors
- Distribution coefficients.

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

Difference between Versions of RESRAD

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

To demonstrate the impact of the change in the code on the outcome of the calculation, we have used the parameter values and pathways reported in the DOE Rocky Flats soil action level report (DOE/EPA/CDPHE 1996) as inputs to RESRAD. The only additional parameter required is the mean annual wind speed. For the purposes of this comparison, we chose to use a value of 4 $m\ s^{-1}$, the value reported by the National Climatic Data Center as the 47 year annual mean for the Denver area (NCDC 1999).

Each scenario, dose level, and radionuclide was evaluated for the impact of this change in the code. The results of the calculations using both version of RESRAD for ^{239}Pu are shown in Table 1. The results for all of the remaining radionuclides are shown in Appendix A to the report. It is important to note that the number of significant digits displayed in this table is for demonstrative purposes only and does not reflect the accuracy of the model.

The table shows that, with just the one change, the soil action levels are much higher with the new version of the code. It is obvious that this single change in the RESRAD code has a large impact on the dose delivered by the resuspension pathway. Therefore, this pathway will be carefully evaluated throughout the remainder of this project to provide accurate and site-specific information and calculations. We describe our approach to the resuspension pathway in more detail later in the report.

Table 1. RESRAD Version 5.61 and 5.82 Calculation Results for ^{239}Pu

Scenario	Dose Level (mrem)	RESRAD Version 5.61 Dose/Source Ratio [mrem (pCi g^{-1})$^{-1}$]	RESRAD Version 5.61 Soil Action Level (pCi g^{-1})	RESRAD Version 5.82 Dose/Source Ratio [mrem (pCi g^{-1})$^{-1}$]	RESRAD Version 5.82 Soil Action Level (pCi g^{-1})
Office worker	15	0.0138	1088	0.00211	7116
Open space	15	0.00151	9906	0.000282	53130
Resident	15	0.0595	252	0.0102	1474
Resident	85	0.0595	1429	0.0102	8351

SENSTIVITY ANALYSIS

To determine the parameters to be examined for uncertainty, we employed a single parameter sensitivity analysis. A single parameter analysis requires that only one parameter be changed at a time to analyze the impact on the solution. This analysis was done earlier in the project for RESRAD Version 5.61 but was completed again using the current version of RESRAD Version 5.82.

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

For this sensitivity analysis, parameter values were allowed to vary by a factor of 10 in either direction (the median value was multiplied and divided by 10) unless the possible range of parameter values would be exceeded. In these cases, RESRAD automatically uses the total possible range of values for the parameter.

The results of this analysis fall into several categories. The parameters of primary importance have been identified as sensitive parameters. Another group of parameters showed limited sensitivity, but in several cases, the values were changed to reflect site-specific conditions. A large fraction of the parameters did not exhibit any sensitivity to change, and these parameters have been identified as such.

Sensitive Parameters

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

- Mean annual wind speed
- Area of the contaminated zone
- Soil-to-plant transfer factors
- Distribution coefficients
- Mass loading.

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

Parameters with Limited Sensitivity

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

231 **DRAFT**

Cover Depth

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

Depth of Soil Mixing Layer

The depth of the soil mixing layer is the depth of surface soil available for resuspension. This value is used to calculate the depth factor, or the fraction of total resuspendible soil that is contaminated. The use of this parameter in RESRAD to calculate depth factor requires that it represent the depth over which contamination is uniformly distributed in the resuspendible layer. In the previous soil action level calculations (DOE/EPA/CDPHE 1996), the values for soil mixing layer and thickness of the contaminated zone are equal, which is not consistent with the definition of either term. *RAC* selected a value of 0.03 m to maintain consistency with the definition. This value has been used in the literature to define the surface or resuspendible soils and is the value defined by Webb et al. (1997) as representative of surface soils at Rocky Flats.

Indoor Dust Filtration

The value of the indoor dust filtration factor represents the fraction of the total outside air contamination that is available indoors. A value of 1 means that the air contamination inside a building is equal to outdoor air contamination. While this is a conservative assumption, *RAC* will not change this value for our independent calculation because of the recognized importance of the inhalation pathway.

Irrigation Water Contamination Fraction

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

Thickness of Contaminated Zone

The thickness of the contaminated zone represents the vertical distance over which radionuclide contamination levels are clearly above background. Changes in this parameter do influence the outcome of the calculation somewhat, but this value has been well characterized at Rocky Flats. The research of Webb et al. (1997) indicates that contamination is distributed over the top 20 cm of soil, with very little movement of that soil within the column over the past 20 years. For this reason, we will treat the parameter deterministically and use a value of 0.2 m (20 cm).

Parameters not Exhibiting Sensitivity

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

External Gamma Shielding Factor

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

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

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

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

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

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

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

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

Initial Concentration of Radionuclides

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

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

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

Radionuclide	Relative concentration (to ^{239}Pu)
^{238}Pu	0.0157
^{239}Pu	1
^{240}Pu	0.186
^{241}Pu	0.994
^{242}Pu	0.00000879
^{241}Am	0.131
^{234}U	0.00819
^{235}U	0.000328
^{238}U	0.00982

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

Plutonium Solubility and Dose Conversion Factors

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

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

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

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

236 **DRAFT**

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

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

[a]ICRP 30 values have been used in RESRAD Versions 5.61 and 5.82
[b]ICRP 71 listed the latest inhalation dose conversion factors [Also given on ICRP CD-ROM (ICRP 1999)]
[c]ICRP 67 listed the latest ingestion dose conversion factors [Also given on ICRP CD-ROM (ICRP 1999)]

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

Remaining Parameters

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

- all of the saturated zone parameters
- all of the uncontaminated zone parameters
- nearly all of the contaminated zone parameters including evapotranspiration coefficient, erosion rate, porosity, conductivity, density, b parameter, precipitation, irrigation rate and mode, and runoff coefficient
- length parallel to the aquifer
- the watershed area
- storage times for food
- mass loading for foliar deposition
- plant contamination fraction
- thickness of the unsaturated uncontaminated zone
- water table drop rate
- well pump intake depth
- well pumping rate.

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

Some of these parameters are a part of the drinking/ground water calculation, and they have no impact on the current soil action level calculation (DOE/EPA/CDPHE 1996) because none of the scenarios include the drinking water pathway. We explore the impact of this pathway in the following section of this report. Table 4 compares the parameter values to be used in the independent calculation to the DOE/EPA/CDPHE values.

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

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

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

The Groundwater/Drinking Water Pathway

Groundwater is an extremely complex pathway (described in Task 2), and *RAC* will not assess it in significant detail in the soil action level project because of the extensive ongoing research and its complexity. We will, however, provide a bounding level, screening calculation for a single scenario (DOE/EPA/CDPHE resident) with contaminated drinking water as a pathway for dose.

For the drinking water pathway, as it will be used in these calculations, the contamination fraction of drinking water is 1.0; that is, 100% of the receptors' drinking water comes from contaminated groundwater. The parameter value for drinking water intake for our rancher scenario, described in the Scenario section, is 730 L y^{-1}, which is a daily intake of 2 L for 365 days. Protecting groundwater resources near their source will protect the resource at farther downgradient locations.

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

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

The increase in dose when the drinking water pathway is included is significant. It is important to understand several things about this calculation and the timing of when the maximum dose occurs. Although ^{241}Am levels are expected to peak around the year 2030 as a result of ingrowth from ^{241}Pu decay, the dose from ^{241}Am does not peak until year 2220. This is a result of the transport of ^{241}Am in the vadose zone. It is also important to understand that the dose imparted by the drinking water pathway does not affect the dose or soil action level for ^{239}Pu or many of the other radionuclides in the study. The dominant pathway for exposure from ^{239}Pu is by resuspension and inhalation, and that pathway will always be most important during the first years after cleanup when radionuclide concentrations have not yet reached the groundwater. Based on the RESRAD conceptual model for subsurface transport and the hydrologic transport parameter used in the simulation, it takes over 200 years for significant concentrations of the americium to reach the groundwater, and, thus, be available in the drinking water.

However, there is much that is not known about the mechanisms by which americium is transported through the soil column and into the aquifer. There is an additional degree of uncertainty about the properties of the aquifer. Studies on the mobility of radionuclides in the Rocky Flats environment do reveal some important information. Both plutonium and americium are strongly adsorbed, limiting their mobility considerably. The distribution coefficient, which describes the partitioning of contaminants between solid and aqueous phase, is quite high for both americium and plutonium at Rocky Flats, indicating a high affinity for the solid phase. Parameters that describe the distribution coefficient, bulk hydrologic properties of the subsurface, and precipitation and infiltration in RESRAD dictate the rate at which radionuclides are transported into the aquifer, and, therefore control the calculation of dose from the drinking water pathway.

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

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

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

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

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

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

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

UNCERTAINTY DISTRIBUTIONS

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

Some environmental parameters are difficult to observe directly, and estimates must be based on inferences from available observations of other presumably correlated quantities. But such an indirect approach usually relies on a model connecting the desired quantity with the ones being measured, and use of the idealized model usually introduces uncertainties of its own. An example relevant to Rocky Flats is resuspension. Factors for wind-driven resuspension have been calculated as the ratio of the air concentration of a contaminant (e.g., Bq of plutonium per cubic meter) divided by the amount of contaminant per square meter of soil (the soil measurement is taken to a depth that is considered resuspendable). A resuspension factor (m^{-1}) is multiplied by a measured soil concentration of a contaminant (e.g., Bq m^{-2}) to predict an airborne concentration of the contaminant (Bq m^{-3}). The implied model is a large source area of soil that is uniformly contaminated and uniform in those properties that affect the mechanisms of resuspension (e.g., ground cover, soil particle size distributions, moisture, depth of the resuspendable layer, and terrain topography). It is also assumed that the resuspension factor represents airborne concentrations that are averaged over a sufficient period to be characteristic of the local meteorological conditions. Such uniformities are seldom available to field studies (or applications), and measurements of factors for wind-driven resuspension range from 10^{-4} to 10^{-11} m^{-1} (Sehmel, 1984). Absent other information, this range is an indication of uncertainty for the local resuspension factor. The resuspension factor for a contaminated location also changes over time as the contaminant migrates downward into soil or undergoes superficial erosion. Anspaugh et al. (1975) and others have made generic characterizations of this temporal trend for plutonium resuspension factors.

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

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

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

Distribution Coefficient

The sensitivity of the drinking water pathway was identified earlier in this report. Much of the uncertainty associated with this pathway results from nonquantifiable uncertainty in the transport of radionuclide contamination into the aquifer, simply from lack of available knowledge about groundwater transport at Rocky Flats. Another very sensitive and quantifiable parameter that affects transport through the vadose zone is the distribution coefficient.

The distribution coefficient, or K_d, defines the ratio at equilibrium of concentration of contaminant per gram of dry soil to the concentration of contaminant per cubic centimeter of liquid. Values for K_d vary greatly with physical and chemical properties of the solid, liquid, and radionuclide. Generally constant for a system under specified conditions, the value for K_d can range over orders of magnitude for different situations. A K_d is a very sensitive parameter in any calculation involving groundwater.

Although we know very little about the actual groundwater transport mechanisms, we can begin to account for a small degree of uncertainty by quantifying ranges of K_d values. Values for K_d have been predicted for the environment around the 903 and Mound Areas, and a range of values has been reported (DOE 1995b). We used these values to create a distribution for K_d for americium, uranium, and plutonium.

The values for the K_d used in the DOE/EPA/CDPHE soil action level document were derived from data reported by Dames and Moore (1984). Dames and Moore reported a range of values for retardation factor, a factor that represents the effects of adsorption on contaminant migration, for sand and clay soils. From these values, K_d was calculated using Equation (3).

$$K_d = \frac{(R-1)n_e}{\rho_b} \tag{3}$$

where
K_d = distribution coefficient (cm^3 g^{-1})
R = retardation factor
n_e = effective porosity of the aquifer
ρ_b = bulk soil density (g cm^{-3}).

Equation (3) was rearranged using an equation in Javandel et al. (1984), who calculated retardation factor using the assumption that the amount of solute adsorbed onto a solid is linearly proportional to the concentration of the solution. For the DOE determination of K_d, the values for n_e and ρ_b were 0.10 and 1.84 g cm^{-3}, respectively. These values were measured for OU-2 and represent site-specific parameters. The Dames and Moore (1984) retardation factor values for sand and clay soils are shown in Table 5 for each radionuclide, along with the associated calculated K_d value calculated using equation (3).

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

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

DOE (1995b) used the midpoint of the ranges shown in Table 5 for americium and plutonium to represent the K_d values for their calculations. Research by Sheppard and Thibault (1990) reviewed a number of studies and produced ranges of K_d values for sand, loam, and clay soils. These ranges are shown in Table 6.

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

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

It is obvious that K_d can vary over several orders of magnitude for even a single soil type, and varies even more across soil types. It is also true that K_d tends to be site-specific.

RAC has created a distribution of K_d values for uranium, plutonium, and americium. Although none of the values extracted directly from the literature are specific to Rocky Flats, the K_d values calculated from the Dames and Moore (1984) data were derived using Rocky Flats information. The midpoints of the K_d values from Table 5 have been maintained as the median estimate for the range of distribution coefficients.

To establish the remaining properties of the distribution, we reviewed the values in Table 6. The range of values in this table clearly illustrates the large range of possible values for K_d. The data from Dames and Moore (1984), used in connection with Rocky Flats information, seems to support lower values for K_d. Additionally, the use of extraordinarily high K_d values would serve only to minimize transport into groundwater. The distribution's lower limit was established using the average of the Sheppard and Thibault (1990) minimum values for sand and clay to represent the 5th percentile of the distribution. The K_d was assigned a lognormal distribution. The geometric mean (median) and geometric standard deviation were developed using the identified literature values and are shown in Table 7.

Table 7. Distributions of K_d Developed for the Independent Calculation (in units of $cm^3 g^{-1}$)

Radionuclide	Geometric mean	Geometric standard deviation
Americium	76	2.52
Plutonium	218	1.16
Uranium	218	3.92

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

Soil-to-Plant Transfer Factors

Soil-to-plant transfer factors quantify that portion of contamination in soil that is transferred to plants via root uptake. In January 1999, the National Council on Radiation Protection and Measurements (NCRP) issued Report No. 129, *Recommended Screening Limits for Contaminated Surface Soil and Review of Factors Relevant to Site-Specific Studies* (NCRP 1999). This screening methodology suggests distributions for soil-to-plant transfer factor that reflect uncertainty resulting from different soil conditions, soil types, and soil chemistry. The values given in Report No. 129 (NCRP 1999) were adapted from values suggested in Report No. 123 (NCRP 1996) with application of uncertainty in the form of a geometric standard deviation. The values with their associated geometric standard deviations are shown in Table 8. These distributions will be used in the independent calculation for Task 5.

Table 8. NCRP Report No. 129 Soil-to-Plant Transfer Factor Values (in units of $Bq\ kg^{-1}$ wet vegetation per $Bg\ kg^{-1}$ dry soil)[a]

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

[a] Source: NCRP (1999).

Area of Contaminated Zone

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

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

To resolve this problem, we will use site data (including air monitoring) to establish relationships between concentrations in air and soil and to use these relationships in applying RESRAD to the site. To carry out this task, it was necessary to define a model of ^{239}Pu concentration in soil as a function of location.

To define this model, we began with a suitable database of observations. We restricted our selection, for the most part, to measurements for which the documentation included the sampling depth and an approximate time when the samples were taken. One series of measurements that did not meet these criteria are discussed below. The sampling depth is important because recent field and theoretical work reported by Webb et al. (1997) established a fractional concentration depth profile for ^{239}Pu at Rocky Flats that can be applied generically to adjust samples taken 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 resuspendable soil and plutonium. The generic profile indicates that essentially all plutonium in the soil at Rocky Flats is currently confined to a depth of 20 cm, with a concentration that decreases with increasing depth. We can then adjust concentrations based on samples taken to depths <20 cm to the 0–3-cm depth by hypothesizing a profile for the sample that is proportional to the standard of Webb et al. (1997). The calculation accounts for plutonium that might have migrated beyond sampling depths less than 20 cm, and a consistent proportion is assigned to the 0–3-cm layer.

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

Webb (1996) summarized estimates of the soil plutonium inventory from several investigations. These estimates are consistent with a regression curve that shows an initial removal of about 40% of the inventory from the 0–3-cm layer in 10 years (Figure 1). The term "regression" refers to a statistical procedure that fits a function or model, which might be visualized as a curve, to a set of data. The procedure can be extended to use the distances of the data points from the fitted curve (called "residuals") to estimate uncertainties in quantities associated with the model.

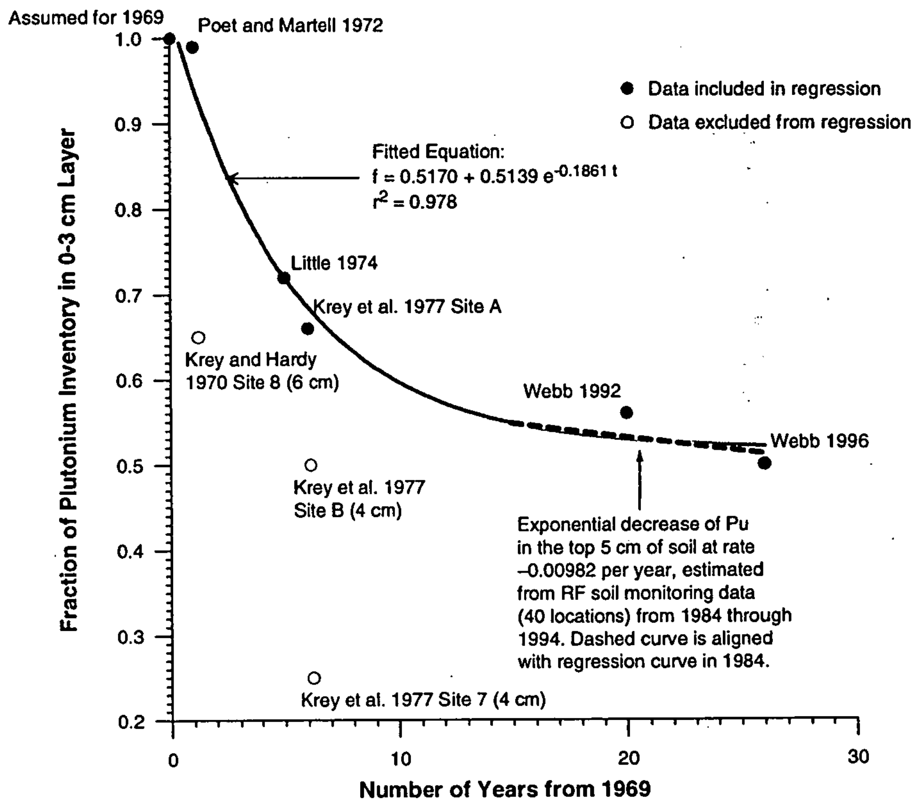

Figure 1. Regression curve fitted to ^{239}Pu data for the 0–3-cm layer of soil at Rocky Flats. The regression was based on data summarized by Webb et al (1996), which are plotted as black circles. The white circles were excluded from the regression. The dashed line represents an estimated exponential removal rate of plutonium from the 0–5-cm layer measured at 40 stations from 1984 through 1994.

This regression curve, presented in Rood (1999), indicates an asymptotic[1] level of about 52% of the initial deposition of plutonium remaining in the 0–3-cm layer. This schedule of decreasing plutonium concentrations is too gradual to be consistent with the conclusions of Krey and Hardy (1970) and with some observations of Krey et al. (1977). Data from some of the locations sampled in these two studies were omitted from the regression because of the apparently inconsistent interpretations. These omitted observations are presented as white-filled circles in Figure 1. Rood (1999) has a fuller discussion of the issues involved. The regression curve in Figure 1 does not explicitly represent details of the mechanisms of transport in the soil. Rather, the form of the function is based on a simple removal model with partial retention.

It is very likely that natural processes continue to remove plutonium from the surface soil, even though the regression curve suggests that the level would never drop below 50% of the

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

initial deposition. *RAC* performed a statistical analysis on samples from the 0–5 cm depth that were collected as part of the Rocky Flats monitoring program. These data were sampled annually from 1984 through 1994 at 40 locations, with distances roughly 1.6 km (1 mi) and 3.2 km (2 mi) from the center of the site and in all directions at intervals of 18°. Using the aggregated data from these locations, we estimated a loss rate of about 1% per year during the 11-year period. A 95% confidence interval for the rate coefficient is -0.0098 ± 0.0182 (-0.0280, 0.0083) per year. Note that this interval includes a segment of nonnegative numbers and thus does not exclude zero loss at the 95% level (however, a 70% confidence interval *would* exclude the zero loss rate). Separate estimates based on the inner and outer circles of sample locations were consistent, giving nearly identical estimates of the rate coefficient.

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

The raw soil concentration data for ^{239}Pu were obtained from two sources: (1) Table I-2 of Appendix I from Ripple et al. (1994), and (2) a computer archive of 1122 results of soil samples, deposited with the CDPHE by M. Iggy Litaor. This archive provided Colorado State Plane (CSP) coordinates (in feet) and activity concentrations (in picocuries per gram) for observations reported by Illsley and Hume (1979). It also provided the CSP coordinates for the 40 locations of the Rocky Flats monitoring series mentioned previously [rings at approximately 1.6 and 3.2 km (1 and 2 mi) from the center of the site, at angular intervals of 18°]. For each of these 40 locations, we averaged the series ^{239}Pu for 1984–1994 for use in our model; 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 archive could not be documented and, therefore, were not used. However, one series, with code numbers PT000–PT124, was considered essential because of the coverage that it provided near the 903 Area. The Rocky Flats sampling protocol specified a sampling depth of 0–5 cm, and we 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. We are also uncertain about the dates of sampling for the PT series. It may be possible to obtain further information on the PT series for Task 5. No other data from this archive were used.

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

Figure 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

DRAFT

250

breakdown gives a fair sense of the spatial distribution of the soil concentrations of ^{239}Pu. Coverage within the plant area and west of the site is relatively thin, and it is unlikely that these areas can be substantially supplemented from other sampling records. Prevailing westerly winds directed most of the attention to areas east of the 903 Area.

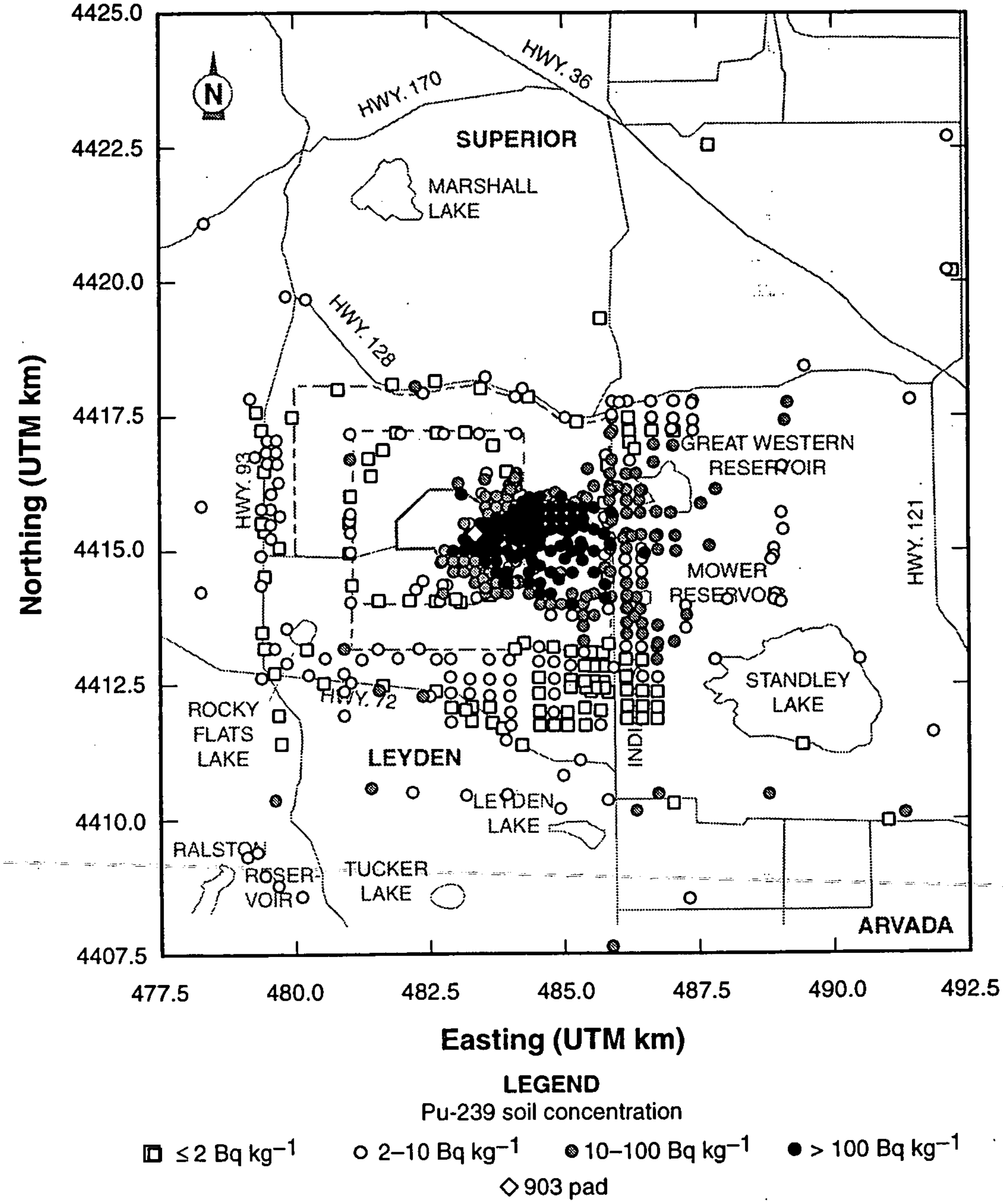

Figure 2. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model. The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), Ripple et al. (1994), and one series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

To be useful, a spatial model of the plutonium concentration in soil must provide estimates for locations not included in the database by means of interpolation. Also, given the considerable spatial variability in the data, the spatial model must provide smoothing. Some efforts have based estimation of contours on kriging methods (Litaor et al. 1995). The *RAC* approach to smoothing was based on the more direct assumption that most of the spatial signal is the result of wind transport of contaminated soil particles from the 903 Area; therefore, a polar[2] representation from this center is reasonable.

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

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[2] The term "polar" means that we represent any location by its distance from a center (or pole) and then angle that the line from the center to the location of interest forms with a specified direction, such as north.

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

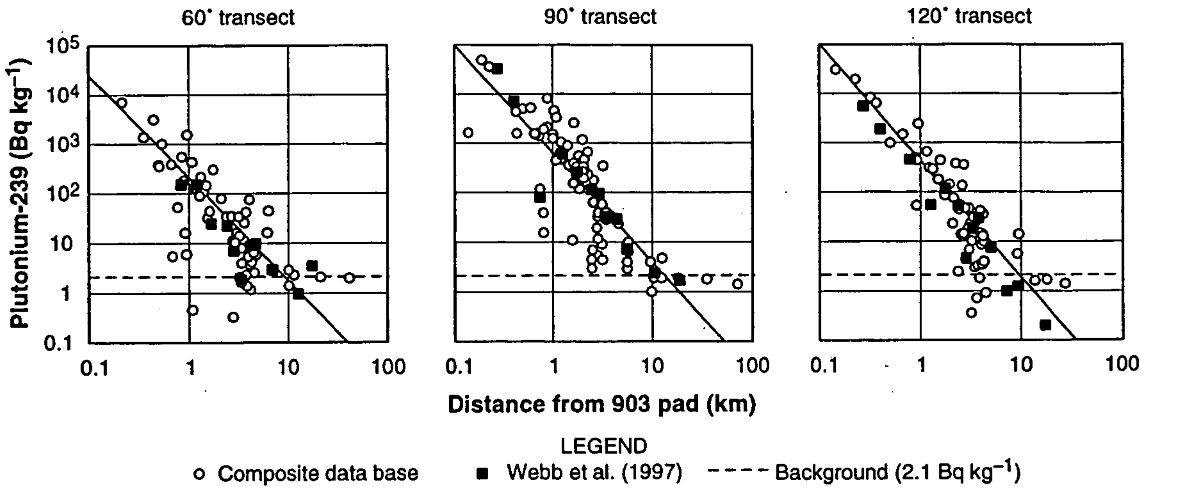

Figure 3. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 Area. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols.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

For the spatial soil model, we fitted a power function to the data within each sector of 22.5°, with centerlines at 0°, 22.5°, 45°, etc. To estimate concentration at points on a sector centerline, the model uses the value of the power function from a point near the 903 Area to the distance at which the power function has the value 2.1 Bq kg^{-1}, which is the estimate of background given by Webb et al. (1997). Beyond this distance, all values are assumed to be background for purposes of the model. Between centerlines of sectors, linear interpolation based on the angle is used to estimate the concentration. For two sectors northwest of the 903 Area (292.5° and 315°), the coverage is inadequate to establish credible power function fits, and the power function for 270° was extrapolated to these two sectors. Contours based on the model are shown in Figure 4.

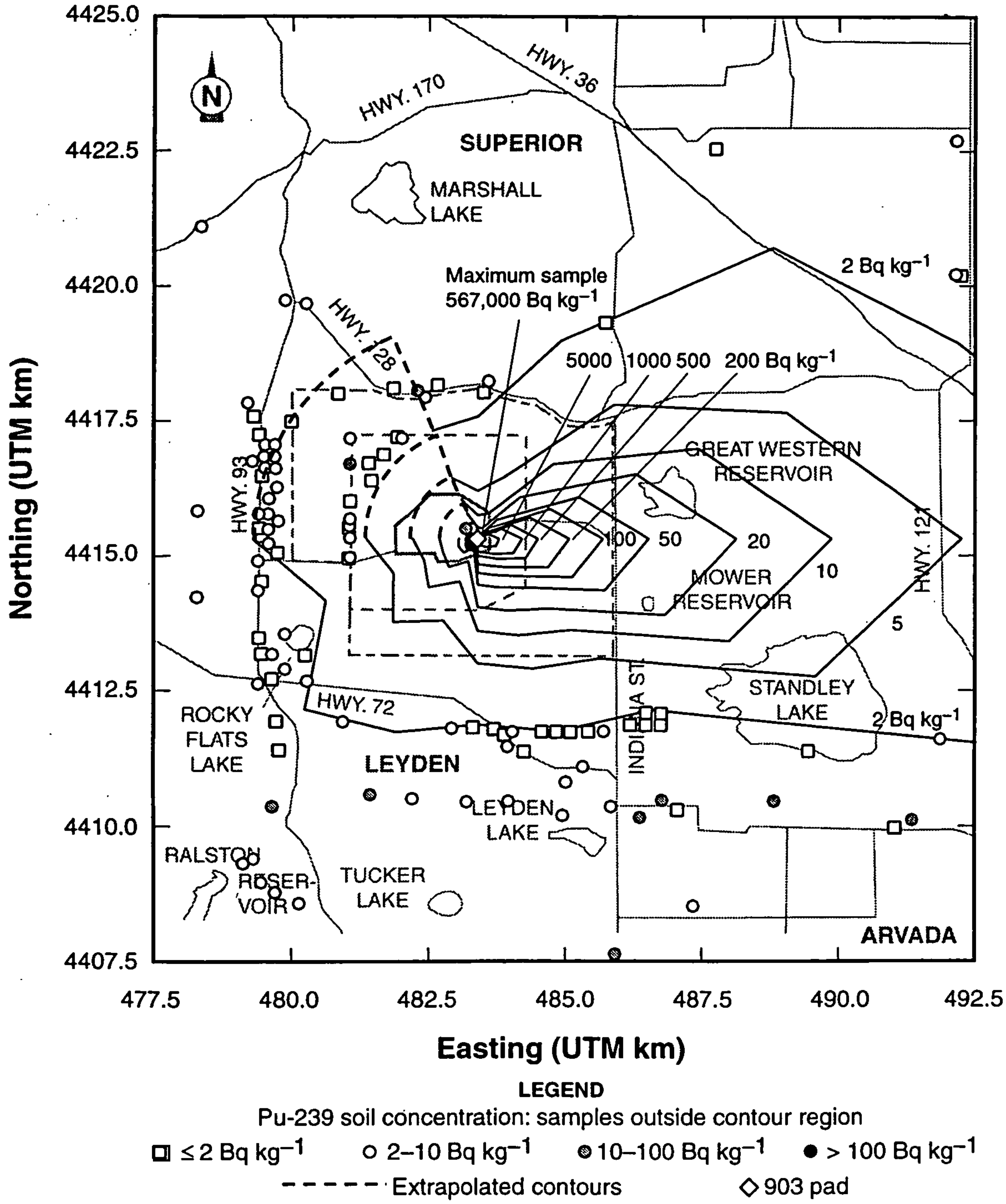

Figure 4. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text.

Dashed lines indicate extrapolation of the two northwest sectors from the 270° sector. Sample locations are shown outside the 2 Bq kg^{-1} contour (approximately background) and within the northwest sectors. Dashed parts of the contours indicate extrapolation where coverage was

254

DRAFT

insufficient for fitting power functions. In these regions and outside the 2 Bq kg^{-1} contour, sample locations were plotted to show that there are some above-background observations where the model would indicate background (2.1 Bq kg^{-1}). However, for purposes of legibility, sample points have been deleted from other regions within the contours.

Although the contours may be considered crude, with an angular resolution no better than the linear interpolation between sectors, they illustrate the considerable variation of the concentrations and the particularly rapid increase as the pad is approached along eastward transects. The model estimates are constrained not to exceed the maximum adjusted sample value (567,000 Bq kg^{-1}), which occurs in the immediate vicinity of the 903 Area. These contours (or any set of contours based on plutonium concentrations in soil at Rocky Flats) cannot be assumed to provide exact partitions according to magnitude. The smoothing and interpolation provided by the model must be kept in mind. The model is not intended to give accurate estimates at specific locations, but rather it provides a basis for integration[4] of resuspension over large areas for calibration..

Mass Loading Factor

RESRAD bases its calculations of resuspension parameters assuming an area of homogeneous contamination. RESRAD defines an area that has homogeneous contamination as any area where all contamination levels are within a factor of 3 of the mean. An area of higher concentration would then be restricted to an area of no greater than 100 m^2. Figure 4 shows that the soil contamination at Rocky Flats is not homogeneous. Clearly, Rocky Flats soil contamination does not fit within the boundaries of this definition needed for RESRAD.

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

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

The tools that *RAC* will use to calibrate resuspension to available air concentration data are being developed and will be described in the Task 5 report. It is important to understand that because of the large degree of inhomogeneity at Rocky Flats, it is difficult to use RESRAD, or most existing assessment programs, to make these calculations. Our methodology provides a way

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

to use the RESRAD tool in combination with available site-specific data to make estimates of resuspension based on actual site conditions.

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

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

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

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

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

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

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

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

Mean Annual Wind Speed

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

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

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

The largest wind events during the period after the 903 Area barrels were cleared and before the area was covered with asphalt were modeled as six discrete wind events. These events produced the largest degree of dust and contamination suspension from the 903 Area. The high wind events were estimated to have been responsible for most of the activity released from the 903 Area. However, high wind speeds also result in greater dispersion, dilution, and depletion within an airborne plume, resulting in lower air concentrations than would be predicted had the same activity been released over a longer period of time and modeled using annual average meteorological data. This is clear if we consider the plutonium concentrations predicted and

257

reported during the Phase I of the Historical Public Exposures Study on Rocky Flats. The average plutonium concentrations for respirable particles near Indiana Street were calculated to be about 0.5 fCi m^{-3} (ChemRisk 1994). This value yields a 6-year time-integrated value of 3.0 fCi-y m^{-3}. The bottom line from these studies is that although the source term for respirable particles calculated in Phase I was about the same as that calculated in Phase II, the total integrated concentration value from the Phase I work is a factor of 2 to 3 higher than that from the Phase II work. Consequently, it appears that that while the discrete events may have contributed to most of the offsite contamination, they do not appear to be as important from an airborne concentration standpoint.

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

SCENARIOS

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

A goal for designing the scenarios in this study is that if the hypothetical individuals are protected by specified dose limits, then it is reasonable to assume that others will be protected. We have given careful consideration to offsite exposures and have designed the scenarios so that if the person living onsite full-time is protected, then the person living offsite also will be protected. The reference scenarios are standards against which levels of radionuclides in the soil at the Rocky Flats site can be measured.

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

RAC is evaluating the three scenarios described in the current soil action level report (DOE/EPA/CDPHE 1996), along with four additional scenarios that we have proposed after numerous discussions with the RSALOP at the monthly soil action level meetings. *RAC* designed specific scenarios during the months of discussion with the Panel and added others at the request and suggestions of the Panel. Ten proposed scenarios were under consideration, and these are briefly described in the Task 2 report. As discussions continued, *RAC* recommended and the Panel agreed that some of the proposed scenarios were very similar to the three current scenarios described in the current Rocky Flats Cleanup Agreement. We consider the three scenarios outlined in the current Rocky Flats Cleanup Agreement as plausible scenarios for the current project.

Table 9 lists the seven scenarios that are currently being evaluated, with the *RAC* scenarios grouped as nonrestrictive and restrictive. Nonrestrictive means that the hypothetical individual has no restriction to the site in terms of time or location. The restrictive scenarios mean that the person's time or location is limited while on the Rocky Flats area. Because the future land use cannot be known with certainty, it is important to include both types of scenarios for evaluation.

Table 9. Summary of Final Scenarios for Evaluation

***RAC* scenarios**		**DOE/EPA/CDPHE scenarios**
Restrictive (Part-time)	Nonrestrictive (Full-time)	
1. Current onsite worker	1. Rancher	1. Residential
	2. Infant of rancher	2. Open space user
	3. Child of rancher	3. Office worker

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

1. Restrictive (part-time): The current onsite industrial worker scenario assumes a person works onsite 8½ hours per day, 5 days per week, 50 weeks a year, or 2100 hours per year. It is assumed that 60% of the worker's time is spent outdoors. The potential pathways of exposure for this person include inhalation, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils. The annual breathing rate is 3700 m^3 per year, based on a time-weighted average of breathing rates and activity levels for the time spent onsite.
2. Nonrestrictive (full-time): The resident rancher scenario assumes future loss of institutional control. The rancher is raising a family, maintaining a garden, and leading an active life at the site, spending 24 hours per day, 365 days per year, or 8760 hours at the site. Of that time, over 40% is spent outdoors. The potential pathways of exposure for this person include inhalation; eating produce from a garden irrigated with some water from a stream on the site, direct soil ingestion from outdoor activities, and direct gamma exposure from the soils and airborne radioactivity. The annual breathing rate is 10,800 m^3 per year, based on a time-weighted average of breathing rates and activity levels.
3. Nonrestrictive (full-time): The child of the rancher family is assumed to be 5 to 17 years of age and onsite 24 hours per day, 365 days per year, or 8760 hours per year. The potential pathways of exposure include inhalation, eating produce from garden irrigated with water from a stream on the site, direct soil ingestion, and gamma exposure from soils and airborne radioactivity.
4. Nonrestrictive (full-time): The infant in rancher family is 0 to 2 years of age and onsite 24 hours per day, 365 days per year, or 8760 hours per year. The infant's potential pathways of exposure include inhalation, some direct soil ingestion from outdoor activities, and direct gamma exposure from soils and airborne radioactivity.

For the soil action level assessment, the scenarios are described and defined by numerous parameters, some much more important than others, as sensitivity analyses have shown. The scenario parameters include breathing rates for various activity levels and ages, soil ingestion rates for children and adults, fraction of time spent indoors and outdoors, and the potential use of or exposure to contaminated water from the area. We have focused primarily on parameter values that provide the greatest impact to the assessment and parameters whose values differ from the RESRAD default values or the current DOE/EPA/CDPHE scenarios (DOE/EPA/CDPHE 1996). Table 10 summarizes the key parameter values for all scenarios.

Table 10. Scenario Parameter Values for DOE and *RAC* Scenarios

	Current DOE/EPA/CDPHE scenarios			*RAC* recommended scenarios			
				Nonrestrictive		Restrictive	
Parameter	Residential	Open space	Office worker	Current site industrial worker	Resident rancher	Infant of rancher (new-born–2 y)	Child of rancher (5–17 y)
Onsite location				Present industrial area	East of present 903 Area	East of present 903 Area	East of present 903 Area
Time on the site (h d^{-1})				8.5	24	24	24
Time on the site (d y^{-1})				250	365	365	365
Time on the site (h y^{-1})	8400	125	2000	2100	8760	8760	8760
Time indoors onsite (h y^{-1})				900	3500	7740	6600
Time indoors onsite (%)	100	100	100	40	60	90	75
Time outdoors onsite (h y^{-1})	0	0	0	1200	5300	860	2100
Time outdoors onsite (%)	0	0	0	60	40	10	25
Breathing rate (m^3 y^{-1})	7000	175	1660	3700	10800	1900	8600
Soil ingestion (g)	0.2 for 350 d	0.1 per visit for 25 visits per y	0.05 for 250 d	0.20 for 250 d	0.20 for 365 d	0.20 for 365 d	0.20 for 365 d
Soil ingestion (g y^{-1})	70	2.5	12.5	50	75	75	75
Irrigation water source	Ground-water	NA[a]	NA	NA	Ground-water	NA	NA
Irrigation rate (m y^{-1})	1	NA	NA	NA	1	NA	NA
Onsite drinking water source	no	no	no	no	Ground-water	NA	NA
Drinking water ingestion (L d^{-1})	NA	NA	NA	NA	2	NA	NA
Drinking water ingestion (L y^{-1})	NA	NA	NA	NA	730	NA	NA
Fraction of contaminated homegrown produce	1	0	0	0	1	0	1
Fruits, vegetables and and grain consumption (kg y^{-1})	40.1	NA	NA	NA	190	NA	240
Leafy vegetables (kg y^{-1})	2.6	NA	NA	NA	64	NA	42

[a] NA = not applicable.

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

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

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

Breathing Rate

We have compiled data from numerous published papers to provide perspective in selecting suitable breathing rates (Table 11). In general, breathing rate studies indicate that gender makes little difference on breathing rates through about age 12. For teens through adulthood, the breathing rate can be 40–50% higher in males than females. There is also age dependency on breathing rates, with adults having breathing rates that are about a factor of 3 higher than for young children. For a person of a given age and gender, the most significant parameter affecting breathing rate is the level of activity; breathing rates can be 15 times higher under maximum work conditions than resting. This activity dependence is important for acute exposure of a few hours, but less important for a continuous chronic exposure (of a year).

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

Based on existing breathing rate studies, *RAC* created distributions of breathing rates for active and sedentary adults, for active and sedentary children, and for active and sedentary infants. Using these distributions and the recommended breakdowns of daily activity for each receptor, we created distributions of scenario breathing rates for each scenario. *RAC* recommended and the panel agreed using the 95th percentile value from these distributions for the scenario breathing rate. Figure 5 shows the distributions for the nonrestrictive scenarios: rancher, child, and infant, and Figure 6 shows the probability distribution for breathing rates for the onsite worker.

Table 11. Summary of Key Breathing Rate Studies Reviewed

Study	Approach	Group	Breathing Rate (L min^{1})
Silverman et al. 1951	Max inspiration and expiration determined for design of respiratory equipment; 1 group; adult males	Male athlete,	
		sitting on bicycle	10.2
		heavy exercise	75
Thompson and Robison 1983	Based on breathing rate at normal body temperature and pressure; 9 age groups; infant through adult; male and female	Adult male	
		resting	8.8
		active	30
Roy and Courtnay 1991	Based on time budgets (hours spent at various activities); 6 age groups; infant through adult; male and female	Adult male resting	7.5
		Adult male, heavy activity	50
Layton 1993	Based on oxygen uptake associated with energy expenditures and metabolism; 7 age groups; infants through adult; male and female	Adult male average	11
		Range based on activity during day	712
Finley et al. 1994	Age-specific distributions for chronic inhalation rates based on Layton 1993	Adults 50th percentile	8.2
		Adults 5th, 95th percentiles	5.8, 11.6
EPA 1997	Deterministic; Outdoor workers (15 men; 5 women)	Light activity	12
		Heavy activity	51
	Outdoor construction workers (19 males)	Light activity	24
		Heavy activity	34

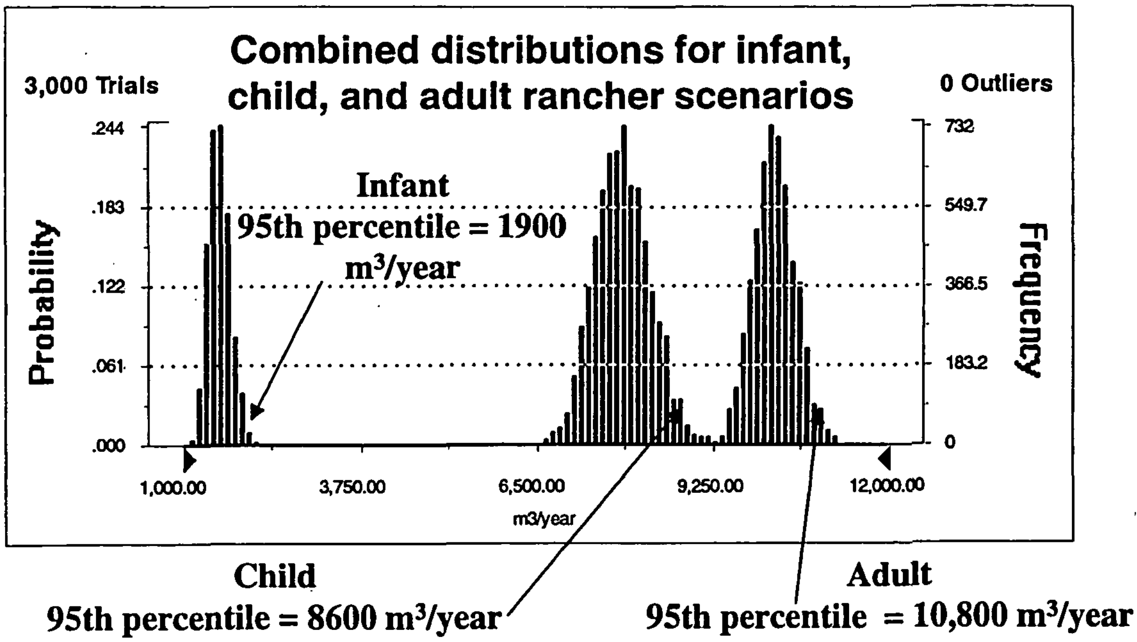

Figure 5. Distributions of breathing rates for the nonrestrictive scenarios: infant, child, and rancher. The 95th percentile of the distribution is shown for each scenario.

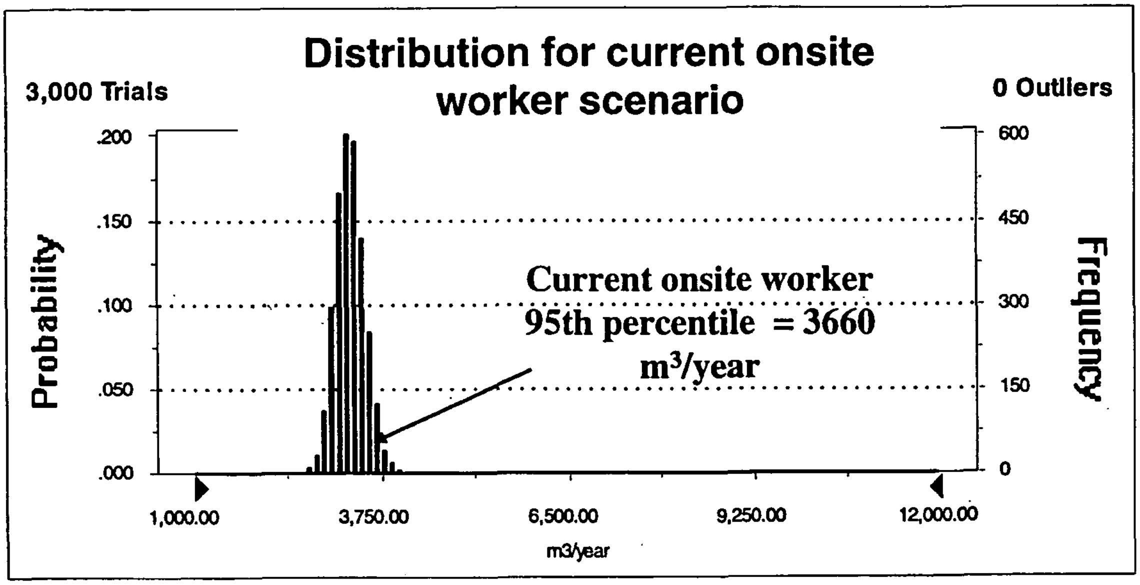

Figure 6. Probability distribution of breathing rate values for the current onsite worker scenario. The 95th percentile of the distribution is $3660\ m^3\ y^{-1}$.

DRAFT

264

Soil Ingestion

Various studies have evaluated the unintentional and intentional ingestion of soil by children and adults. Table 12 lists the studies used in selecting the soil ingestion rate for our scenarios. The table summarizes the approach used in assessing ingestion in each study and the geometric mean and geometric standard deviation for those studies. In 1984, the Centers for Disease Control (CDC) estimated age-specific soil ingestion at about 10 g d^{-1} based on observations of behaviors of children of 1 to 4 years of age (Kimbrough et al. 1984). In 1986, one of the first quantitative assessments of human soil ingestion was carried out using tracer elements in the soil like aluminum, silicon, titanium (Binder et al. 1986). In 1990, Calabrese et al. (1990) studied soil ingestion rates in adults and children using a mass balance approach and more controlled procedures. Simon (1998) developed scenarios based on an extensive review of the literature. The scenarios applicable to this current soil action level study are for a rural lifestyle with homes in a sparsely vegetated area, similar to the Rocky Flats area. Simon assumed a lognormal distribution for inadvertent soil ingestion for adults with a geometric mean of 0.2 and a geometric standard deviation of 3.2. For children living this lifestyle, the geometric mean is 0.2 g d^{-1}, with a geometric standard deviation of 4.2 to develop a distribution of values, and a median estimate of 0.2 (which would give 5th and 95th percentile values of 0.02 g d^{-1} and 2 g d^{-1}, respectively).

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

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

Table 12. Summary of Soil Ingestion Studies Reviewed

Study	Approach	Soil ingestion ($g\ d^{-1}$) Geometric mean	Geometric Std Dev
Simon 1998 NCRP Report 129 (NCRP 1999)	Scenarios based on literature review: Rural lifestyle (w/homes)- sparsely vegetated		
	Lognormal adults(inadvertent)	0.2	3.2
	Lognormal children(inadvertent)	0.2	4.2
Thompson and Burmaster, 1991 (Reanalysis of Binder et al. 1986)	Lognormal distribution (Children)	0.06	2.8
Stanek and Calabrese 1995 (Reanalysis of Calabrese et al. 1990)	Range of median soil ingestion of 64 children over 365 days	0.001 to 0.10	
	Median of daily average soil ingestion of 64 children:	0.075	
	Range of upper 95% soil ingestion estimates	0.001 to 5.3	
	Median upper 95% soil ingestion estimate of 64 children over 365 days:	0.25	
Calabrese et al. 1990 (children)	Distribution percentiles Median (5th, 95th percentiles)	0.02 (0, 1.2)	
Thompson and Burmaster, 1991 (included geophasic children)	Distribution percentiles Median (5th, 95th percentiles)	0.06 (0.01, 9)	
Kimbrough et al. 1984 (children)	Deterministic Mean (low, high)	0.1 (0.05 - 5)	
Hawley 1985 (adults)	Deterministic (average estimate)	0.06 - 0.07	
EPA1994 (adults)	Deterministic (conservative)	0.1	
NCRP Report 123 (NCRP 1996)	Deterministic (conservative)	0.25	

266 **DRAFT**

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

We reviewed various published soil ingestion studies and fit a probability distribution to the data from these studies (NCRP 1999; Simon 1998; Stanek and Calabrese 1995; Thompson and Burmaster 1991; Calabrese et al. 1990). We then looked at how deterministic values from other studies fit into the probability distribution (Kimbrough et al. 1984; EPA 1997; NCRP 1996; Hawley 1985). Figure 7 shows the probability distribution for the soil ingestion studies. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown. As stated above, we are using the 50th percentile of this distribution (0.2 g d^{-1}) as the daily soil ingestion rate for our scenarios.

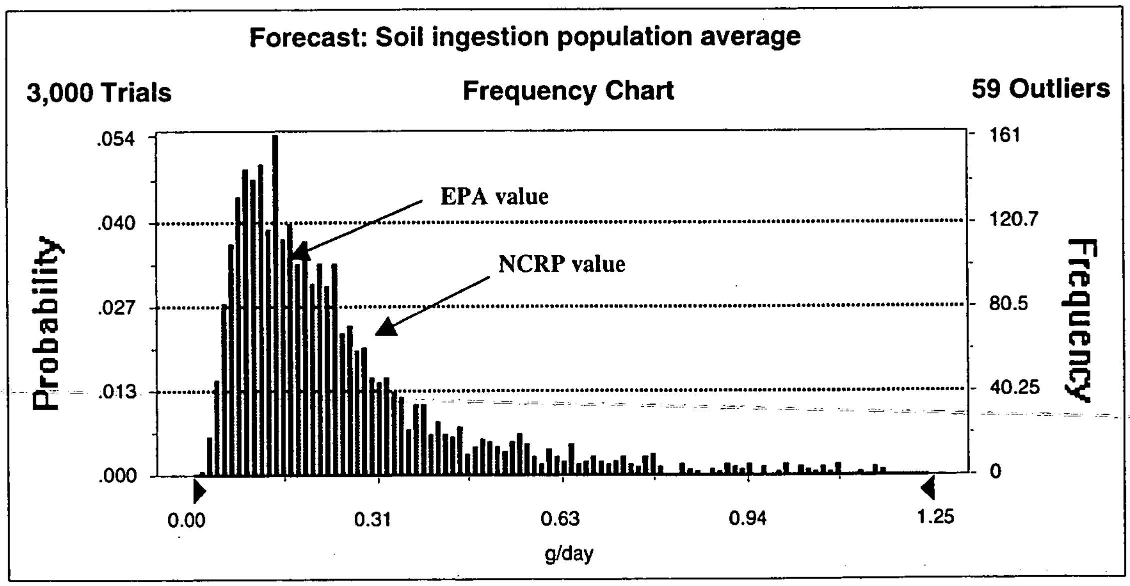

Figure 7. Frequency distribution of soil ingestion values from CrystalBall©. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown.

Groundwater as Irrigation and Drinking Water Source

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

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

Drinking Water Intake

We recommend a drinking water intake of 2 L d^{1} for the adult rancher scenario based on regulatory guidance from the EPA (1989, 1997) and from other studies (Layton et al. 1993). The current DOE/EPA/CDPHE scenarios do not include drinking water as a potential pathway. The RESRAD default value for drinking water ingestion is 510 L y^{1}.

Fruits, Vegetables, and Grain Consumption

We recommend an annual consumption rate for fruits, nonleafy vegetables, and grains of 190 kg y^{1} for the rancher scenario, 200 kg y^{-1} for the infant scenario, and 240 kg y^{-1} for the child scenario (NCRP 1999). Consumption of leafy vegetables is assessed separately in RESRAD. For the *RAC* scenarios we assume the consumption of leafy vegetables at the rate of 64 kg y^{-1} for the rancher, 26 kg y^{-1} for the infant, and 42 kg y^{-1} for the child scenarios. The current DOE/EPA/CDPHE scenarios assume 40.1 kg y^{-1} of vegetables, fruits, and grains and 2.6 kg y^{-1} of leafy vegetables. The RESRAD default values for these parameters are 160 kg y^{-1} for fruits, nonleafy vegetables, and grains, and 14 kg y^{-1} for leafy vegetables.

268 **DRAFT**

CONCLUSIONS

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

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

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APPENDIX A:

COMPARISON OF RESULTS BETWEEN VERSIONS 5.61 AND 5.82 OF RESRAD

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APPENDIX A: COMPARISON OF RESULTS BETWEEN VERSION 5.61 AND 5.82 OF RESRAD

The following table shows the difference between the dose to source ratio and the soil action levels for the two versions of RESRAD described in the main text of the report. All radionuclide results are shown for all DOE scenarios.

			RESRAD Version 5.61		RESRAD Version 5.82	
Scenario	Dose Level (mrem)	Radionuclide	Dose/Source Ratio [mrem $(pCi\ g^{-1})^{-1}$]	Soil Action Level $(pCi\ g^{-1})$	Dose/Source Ratio [mrem $(pCi\ g^{-1})^{-1}$]	Soil Action Level $(pCi\ g^{-1})$
Office Worker	15	^{241}Am	.0718	209	.0548	273
		^{238}Pu	.0129	1164	.00197	7631
		^{239}Pu	.0138	1088	.00211	7116
		^{240}Pu	.0138	1089	.00210	7197
		^{241}Pu	.00192	7801	.0000354	10250
		^{242}Pu	.0131	1145	.00198	7577
		^{234}U	.00922	1627	.00421	3563
		^{235}U	.133	113	.128	117
		^{238}U	.0296	506	.0252	596
Open Space	15	^{241}Am	.0117	1283	.00992	1513
		^{238}Pu	.00142	10580	.000265	56550
		^{239}Pu	.00151	9906	.000282	53130
		^{240}Pu	.00151	9919	.000280	53500
		^{241}Pu	.000312	48020	.00000457	56810
		^{242}Pu	.00144	10430	.000266	56310
		^{234}U	.00130	11500	.000777	19310
		^{235}U	.0114	1314	.0109	1373
		^{238}U	.00295	5079	.00248	6044
Resident	15	^{241}Am	.395	38	.324	46
		^{238}Pu	.0555	270	.00951	1577
		^{239}Pu	.0595	252	.0102	1474
		^{240}Pu	.0593	253	.0101	1490
		^{241}Pu	.00429	3499	.00329	4553
		^{242}Pu	.0564	266	.00956	1569
		^{234}U	.0489	307	.0278	541
		^{235}U	.625	24	.609	25
		^{238}U	.146	103	.126	119
Resident	85	^{241}Am	.395	215	.324	262
		^{238}Pu	.0556	1529	.00951	8935
		^{239}Pu	.0595	1429	.0102	8351
		^{240}Pu	.0594	1432	.0101	8444
		^{241}Pu	.00429	19830	.00329	25800
		^{242}Pu	.0564	1506	.00956	8890
		^{234}U	.0489	1738	.0277	3066
		^{235}U	.630	135	.609	139
		^{238}U	.145	586	.126	674

APPENDIX B:
JOINT FREQUENCY TABLES FOR FIVE YEAR AVERAGE ROCKY FLATS WIND SPEEDS

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Stability Class A

Fraction of total meteorological data in stability class = 0.0133

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

Stability Class B

Fraction of total meteorological data in stability class = 0.1255

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

Stability Class C

Fraction of total meteorological data in stability class = 0.1734

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

Stability Class D

Fraction of total meteorological data in stability class = 0.2752

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

218

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Stability Class E

Fraction of total meteorological data in stability class = 0.1734

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

Stability Class F

Fraction of total meteorological data in stability class = 0.2381

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

Composite of all Stability classes								
	Windspeed (m s^{-1})						Direction Fractional Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00149	0.00890	0.01309	0.00670	6.99008	5.99936	0.03149	4.23623
NNE	0.00269	0.01779	0.01700	0.00569	2.99968	0	0.04349	3.53334
NE	0.00369	0.02089	0.01159	0.00170	2.00896	0	0.03809	2.93183
ENE	0.00519	0.02159	0.00720	0.00059	0	0	0.03460	2.51795
E	0.00589	0.02730	0.01110	0.00239	1.99644	0	0.04690	2.78689
ESE	0.00580	0.02979	0.01169	0.00099	0	0	0.04829	2.64085
SE	0.00500	0.02849	0.01929	0.00260	3.01468	1.00572	0.05580	3.04324
SSE	0.00489	0.02819	0.02340	0.00549	4.99612	1.99644	0.06269	3.32161
S	0.00650	0.02980	0.02669	0.00730	7.00508	9.9072E	0.07109	3.36908
SSW	0.00579	0.02630	0.01999	0.00399	0.00039	9.9072E	0.05659	3.15415
SW	0.00580	0.02710	0.02630	0.00549	5.00864	9.9072E	0.06530	3.32627
WSW	0.00649	0.02910	0.03290	0.01259	0.00240	0.00079	0.08429	3.82823
W	0.00659	0.03579	0.02629	0.02040	0.00879	0.00939	0.10729	4.83505
WNW	0.00509	0.02829	0.02269	0.03380	0.01740	0.01560	0.12290	5.90207
NW	0.00389	0.02340	0.02559	0.01720	0.00510	0.00169	0.07689	4.47467
NNW	0.00309	0.01609	0.02540	0.00799	5.00864	0	0.05310	3.80132
Totals	0.07799	0.39889	0.32029	0.13499	0.03800	0.02869	0.99888	3.87038

280 **DRAFT**

RAC RESPONSES TO PEER REVIEWER COMMENTS

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

August 1999

"Setting the standard in environmental health"

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281

Responses to Reviewers' Comments On RSAL Task 3 Report

Upon reading the reviewer comments for the Task 3 report, *RAC* noted instances where we may not have clearly described the boundaries of the soil action level project, and, in particular, the limitations of this Task 3 report. It seems appropriate to remind the panel and ourselves of some of these constraints prior to addressing the reviewer comments. Additionally, much of the discourse that occurs at panel meetings guides the decisions made by *RAC* with regard to parameters, scenarios, and general project course. These conversations are generally unknown to the reviewers, and though they facilitate *RAC* and the panel's understanding of where this project is headed, they can be difficult to capture in a technical report, other than to state that discussions occurred and decisions were made based on these discussions. *RAC* has tried to summarize much of this information in the reports as possible, but it is sometimes difficult to convey the full explanation to the reviewers. When a situation like this is evident in the reviewer comments, we will point out the source of the misunderstanding and do our best to make the report as clear as possible with regard to the decisions made.

It is important to remember that the soil action level project is severely limited by budget and time constraints. In light of these constraints, we have endeavored to do the best science possible, and realize that, at some point beyond the scope of this project, further enhancements to this work may be desirable.

The goal of Task 3 was to identify parameters in RESRAD, based on that model's selection in Task 2, whose values, when changed, impacted the outcome of the soil action level calculation in a significant way. We were forced to streamline our efforts in this area, and not use resources to determine either uncertainty or alternate values for parameters that were not sensitive to change. Only obvious parameters from this category that justified change were adapted. Naturally, the parameters that emerged as obvious were the ones closest to *RAC*'s previous experience with resuspension and surface soil properties. Given more time and resources, there were a number of parameters that might have been subject to some degree of change and/or development of uncertainty based on a more thorough literature review and some necessary professional judgement.

In the context of this project and for the benefit of the panel, we have used published numerical data for quantification of uncertainty, whenever possible. As a result, we have tried to restrict widespread application of professional judgement in the area of quantifying uncertainty. This approach has proven to be confusing in other areas of the project.

This reminder of the goals and limitations of Task 3 and the project as a whole provides a background for responding to the comments of the reviewers.

PEER REVIEWERS

Reviewer A

General Comments

The general effort to incorporate as much site-specific information as possible into the RESRAD code is appropriate and to be applauded.

A number of the parameter assumptions adopted in the report are questionable to this reviewer. Some that are questionable are discussed under specific comments. It is not evident that the parameter assumptions are based on the most thorough and critical review of the existing literature.

It is recommended that some experts (for example, Greg Choppin, Florida State on K_d assumptions) be consulted on the reasonableness of some of the parameter values and their uncertainties.

This review was perhaps less than adequate because travel commitments of the reviewer precluded a full, comprehensive review with detailed recommendations for additional sources of information.

Specific Comments

The K_d of 218 cm^3 g^{-1} for Pu seems at least two orders of magnitude too low. This value would not be consistent with the characteristics of Rocky Flats Soil, which is high in clay, nor with the observed behavior of Pu in the Rocky Flats environment. Furthermore, the GSD of 1.16 is way to low. This implies that the uncertainty on the value is quite small, which it is not. Secondly, ground and perhaps surface water are the main things this parameter would affect, so I am puzzled as to why this parameter was sensitive. However, a K_d of only ~200 would allow fairly rapid surface depletion of Pu, which would reduce resuspension. This could explain the sensitivity, although this was not explained, unless I missed it. However, this is even more confusing, since I think the approach is to use measured mass loading in any case to derive the inhalation exposures.

The K_d value for Am is also too small, I believe, but the GSD value seems reasonable.

I'm not happy with the way these values were derived in any case. Apparently, they trace back to retardation factors developed by Dames & Moore. I think much more can and should be done to come up with and justify more reasonable K_d values.

We are reviewing Actinide Migration Panel studies to further enhance our K_d evaluation.

The soil to plant transfer factors were listed as "sensitive" parameters. First of all, I am a bit surprised by this, since one would expect food chain exposure to be a very small fraction of the inhalation exposure. This needs some explanation. Secondly, it is not clear whether these values represent strictly root uptake, or a combination of root uptake as well as dust loading. If they represent strictly root uptake, I think there are data to indicate about an order of magnitude smaller value for Pu at least. If the values represent root uptake plus dust loading, then the values

283

are too small, by roughly an order of magnitude. I'm not certain how the computations are handled in RESRAD, but this needs to be explained.

Upon reviewing data for this report and the project in general, the recommendations of NCRP Report No. 129 were explored. It was decided early in the project to include the distributions for soil-to-plant transfer factors because of the inclusion of the agricultural pathway for some of the scenarios. These parameters are, however, only moderately sensitive to change, as pointed out by the reviewer. They were mistakenly included in the "most sensitive parameters" section because we planned to include a distribution. We will move the discussion on these parameters to the section titled "Parameters with Limited Sensitivity".

In RESRAD, the soil-to-plant transfer factors represent only that fraction of contamination that is transferred to plants via root uptake; the dust loading calculations are handled through the use of a mass loading for foliar deposition parameter and calculation.

The area of contamination is listed as 40,000 m^2. I think this is too small, but apparently, the computations are going to somehow use actual soil data in a spatial sense. It is not clear to me how this will be done, and whether or not the assumption of a particular area is even important, if this is to be handled in some spatial scheme that is not normally tackled by RESRAD.

The 40,000 m^2 area listed was the area used in the previous DOE calculations. The current calculations will derive an area based upon scenario assumptions and use this area and the contamination associated with it to develop air concentrations as indicated by the available monitoring data. This evaluation will be appended by a modifying factor, which will attempt to account for a situation in which groundcover is eliminated, making contaminated soil much more available for resuspension.

The mass loading estimate of 2.6×10^{-5} is reasonable for most rural locations. However, why is this even important to debate here if actual soil loadings are to be used? If actual soil loadings are to be used, what soil concentrations for the radionuclides are to be used, given that the source of dust would most likely be quite general?

The mass loading factor shown in the text is again the factor used in the previous calculations. Current calculations will utilize available information to develop actual soil loadings. The radionuclide concentrations in the soil currently, described in the section titled "Initial Concentrations of Radionuclides," will be used for the contaminated soil. Additional soil contributing to the soil loading profile will be assumed to result from uncontaminated soils in the upwind fetch.

The statement on page vii "High wind also results in lower air concentration than would be expected if the same material was dispersed over a longer period of time during average wind speed conditions." needs some explanation and documentation. This could be true, unless average wind speeds were insufficient to cause any measurable resuspension, due to good vegetation cover.

This statement comes from results of the dose reconstruction study done at Rocky Flats. This study predicted that although high winds likely resulted in a large degree of soil movement, the dispersion of this material was so great that the concentration of contamination in air was significantly less than that which resulted from average wind speed conditions. This dispersion effect is magnified close to the source, which is the location of the receptor in this study. The statement in the executive summary is expanded in the section of the report dealing with average wind speeds.

The value suggested for the depth of soil available for resuspension, namely 3 cm, seems way too high to me. Most studies have indicated that on a time scale of < 1 year or so, only a couple of mm are likely to be available for resuspension, unless the site is highly erodable due to overgrazing, lack of vegetation or mechanical disturbance.

The depth of soil available for resuspension represents the layer of soil within which contamination in uniform. The selection of this value was dictated primarily by the available soil data, most of which represented area to that depth. Although it would be desirable to represent the contamination in a shallower layer, the data available to us make it difficult to estimate contamination to any other depth. The research of Webb et al. (1997) showed that throughout the top 3 cm, the contamination is primarily uniform, with perhaps a slight dip in the contamination at the lower depths. Webb et al. also provide a means to convert contamination profiles at other depths to the 3 cm depth. Since we are constrained to this depth by the available data, we must use it for the depth of soil available for contamination. As erosion progresses, uniformly contaminated soil from the lower part of this 3 cm will likely be exposed to resuspension. We will incorporate a better description of this parameter in the final version of the report.

The assumption that the irrigation contamination fraction is 1.0, seems unreasonable. This needs more justification, especially since groundwater does not seem contaminated. What about surface water on the other hand? Is this included in the model?

As a part of an agreement reached with the panel overseeing this study, we agreed to include contaminated groundwater as a possible pathway for exposure. Since one of our exposure scenarios is a residential rancher, allowing irrigation water to be contaminated was an important possible pathway. Because we assume that the source of the irrigation water is directly from a groundwater well located beneath the site, the contamination fraction is set at a value of 1.0 to make the irrigation water as contaminated as the groundwater. The groundwater pathway analysis is included only as a screening calculation, to show the possible effects of groundwater at the site and to direct future studies.

285

Reviewer B

Review Summary

The content of the above named report is focused on a discussion of *RAC*'s chosen values for model parameters, the assumptions used to justify the choice of those parameter values, and on a sensitivity analysis of the soil action level calculation.

This report was organized in a reasonable way and sufficient detail was presented for most parameters. The Executive Summary seems rather long for a report of this length. Many of the chosen values for parameters seem reasonable, others in my view are not credible; each are discussed in the remainder of this review.

Being that the purpose of the report was to present the results of a sensitivity analysis (stated on p. v and p. 1), the report was not completely successful because the method of conducting the sensitivity analysis did not allow for the analysis to reflect the range of sampled values from each distribution (see my comment #8 below).

Other detailed comments are found below.

Detailed Comments

p. vi. The first of several times, it is stated that "soil-to-plant transfer factors quantify that portion of contamination in soil that is transferred to plants via root uptake". This is not a correct interpretation of soil-to-plant transfer factors. These factors represent the fraction of the concentration of the soil within the root-zone of the plant that is observed in plants – also on a concentration basis. Because of the much smaller mass of the plant relative to soil, it is *not* the fractional transfer of the soil inventory. Such incorrect statements appear on p. 19 and possibly elsewhere in the report.

Although the authors certainly had the correct definition in mind when writing the report, we thank this reviewer for noting this inconsistency with the appropriate definition. In an attempt to be as clear as possible for the majority of the audience of this report, we left open a door for misinterpretation of our definition. We will clarify this definition in the final report.

p. vii. It is noted that that RAC will use a 5-year average wind speed, etc. for modeling resuspension, but a few sentences later comments that a "distribution of wind speed values" will be used. It would be useful to explain here very briefly if the distribution discussed is a model of the uncertainty in the average or if not, to clarify the distribution.

We intend only to use the 5-year average STability ARray (STAR) met data for modeling the resuspension. Although we examined the change in the average from year to year, we discovered very little fluctuation in annual average. We intend to remove the statement from the executive summary that indicates the use of a distribution.

In the Executive Summary and elsewhere in the report (for example, see beginning sentence of Executive Summary-Scenarios), it states "The Task 3 report describes....". At this point, I had to look back at the cover to reaffirm that I was reading the Task 3 report. It would be better to

286

state, "This report describes...", thus, eliminating any confusion about which report is being referenced. This occurs elsewhere in the report.

We include statements like this for clarity, since we refer throughout the document to a variety of reports. We appreciate this comment, and will make the discussion as clear as possible in the final report.

On the top of p. ix, the authors state "RAC created distributions..." I suggest that the preferred technical language would be "RAC defined distributions...". This language appears on p. 18 and possibly elsewhere in the report.

We appreciate this comment and will incorporate this language.

p.2, last paragraph. Rephrase: "It is obvious that this single change in the RESRAD code has a large impact on the dose delivered by the resuspension pathway" *to* "It is obvious that this single change in the RESRAD code predicts a significantly different dose via the resuspension pathway".

We appreciate this comment and will incorporate this language.

I note from Table 1 that RESRAD Version 5.82 predicts a soil action level about 6-fold greater than does version 5.61. Such a dramatic change between what seem to be similar versions of the code (based on their version numbers) raises questions about the scientific basis for the resuspension calculation as well as other pathways in the code. It is impossible for external reviewers such as I to judge the validity of the code before or after such changes. This point is raised here as a precautionary flag to RAC and RSALOP that the technical basis for calculations in the code needs to be continually scrutinized as each version change is made.

We recognize this dramatic change as well. Although a perusal of the RESRAD documentation seems to indicate that the change in the code is warranted scientifically, we decided to utilize site-specific data in our evaluation of resuspension and create an external resuspension model rather than to use the one internal to RESRAD V. 5.82.

P. 4 notes that "a single parameter uncertainty analysis *requires* [my emphasis] that only one parameter be changed at a time." This is an overstatement in my view and sounds as if the ends justified the means. It would be more accurate to state that "a single parameter uncertainty analysis *is defined by changing* only one parameter be changed at a time." Moreover, single parameter uncertainty analyses are not regarded as state-of-the-art; I think that fact should also be given some note in the report. State-of-the-art sensitivity analyses vary all parameters simultaneously and rank the sensitivity of each parameter based on the fraction of the output variance contributed by each parameter. Such techniques are generally more difficult to implement. Techniques of lesser sophistication, such as that available in RESRAD, can be used, but their limitations should be noted.

The authors and the panel recognize that this sensitivity analysis is not state-of-the-art. A more rigorous analysis will be completed for Task 5 of this project, using the distributions defined in this report. However, the single parameter analysis was required

287

for this portion of the project, and, in fact, reveals the information about each parameter that we were looking for – how important is each parameter in the calculation of soil action levels and doses?

The metric by which sensitivity was judged was not mentioned in the report. Was it the absolute or relative change in the output?

Due to many comments regarding the sensitivity analysis, we will make efforts to more carefully describe it in the final version of the report.

Given that RAC has discussed the necessity of a dynamic (time-dependent) model for determining soil action levels, has the sensitivity to the set of parameters been determined over (future) time?

Part of the reason RAC prefers the modeling approach outlined here is because of the ability to evaluate soil action levels under a variety of conditions (e.g. current, remediated, future catastrophic event) that may be present at future times.

P. 4, The sensitivity analysis was not performed appropriately to determine the sensitivity of the model to the parameters and their specified distributions. The third paragraph states the "parameter values were allowed to vary by a factor of 10 in either direction." Sensitivity analysis is intended to show the sensitivity of the output variable to both the mathematical structure of the model and the legitimate range of variation of parameters. By presetting all parameters to the same degree of variation (10x in either direction), the sensitivity of the model to the variability of the parameter is lost. Only the sensitivity to the model structure is retained. Thus, from the results presented, it is not easy (or maybe not even possible) to see the true sensitivity of the model to each parameter. RAC should consider redoing the analysis.

***RAC* will not endeavor to redo the sensitivity analysis. We recognize that we are not evaluating true sensitivity to variability (or change) in the parameter, but rather to the model output's sensitivity to changes in the parameter value. This is, however, the sensitivity we were seeking to evaluate at this juncture of the project. Quantifiable variability in the parameters is designated in this report; sensitivity to this variability is a part of the final task of this project.**

Depth of Soil Mixing Layer (p. 5): RAC has selected the depth of 0.03 m (3 cm) as the depth of soil available for resuspension. This is certainly a better choice than the thickness of the contaminated zone (over which the concentration may vary substantially).

We agree and appreciate this reviewers comment.

Indoor dust filtration (p. 5): The definition of this is poorly stated in the same way that the soil-to-plant transfer was poorly stated. In the two opening sentences, "contamination" should be changed to "concentration" because "contamination" is too vague and could imply inventory, which is definitely not equal to concentration (since the volume of the house is much smaller than the volume of the atmosphere!).

Again, we appreciate the suggestion for clarification of our definition and will make the appropriate adjustments.

Moreover, RAC assigns an equal value to the indoor air concentration, notes it is a conservative assumption, assigns no uncertainty, and states *a priori* that they will not change the value. This is the first of several locations, where RAC fails to produce a credible uncertainty analysis due to the assumptions they make. The noteworthy problems in their method are as follows. 1) An uncertainty analysis should determine credible bounds around a realistic central value of the model output (in this case, the soil screening level). It is impossible to determine credible bounds on the output parameter when some input parameters are set to "conservative" values (in other words, higher than likely) as these parameters will skew the entire result toward larger and unrealistic values. 2) Assigning no uncertainty to a parameter is the same as stating confidence in the value. No one could possibly assert confidence in the assumption that indoor concentrations equal (exclusively and without variation) the outdoor concentration. 3) An uncertainty analysis *requires* (and *requires* is used correctly here) that the assessor be unbiased in choosing parameter values and be impartial to changing those values, as is dictated by the science. This is clearly not the case here as RAC as chosen to purposely maximize the pathway (that is the meaning of choosing *conservative* values) in the interest of not underestimating the inhalation dose.

***RAC* appreciates this reviewer's comments about uncertainty analyses, but we do feel it is important to point out a few key elements of this project that dictate the direction we must take. First of all, it is important to remember that the purpose of Task 3 was to evaluate the input assumptions assigned to RESRAD parameters as they were used in the prior analysis (DOE/EPA/CDPHE 1996). This boundary condition on the analysis was put in place because of two important factors: 1) The panel was interested in knowing how the values selected for the previous analysis affected the calculation, and 2) the limitations on this project prevent us from doing an analysis such as that suggested by the reviewer. In light of these two factors, the sensitivity analysis was set up in such a way as to maximize our resources and minimize our effort on parameters for which a credible value had been chosen for the previous analysis.**

It does not follow that assigning no uncertainty to a parameter is the same as stating confidence in the value. What it means is that, under the limitations of this project, we saw no reason to change the parameter from its previous value. In the case of the indoor air concentration, the value used in the previous analysis, 1.0, was determined to be reasonable given that we know very little about the future conditions at the site.

Based on the comments of a number of reviewers, however, we plan to examine a distribution of values for this parameter.

Irrigation Water Contamination Fraction (p. 5): The same comments as Indoor dust filtration apply here.

This factor was discussed in the set of review comments from Reviewer A.

External Gamma Shielding Factor (p.6): Equation 1 describes a *weighted shielding factor* and not an *occupancy factor* (which is the fractional time spent indoors). I don't know whether

290

RESRAD is responsible for such poor names for variables or if it is RAC's choice; either way, it is incorrect.

The variable name "occupancy factor" is one that was assigned by the RESRAD designers and is cited in the documentation for the code. We will continue to use it in our text.

What is the uncertainty assigned to the shielding factor of 0.7 chosen by RAC?

We assigned no uncertainty to the shielding factor, as it was a parameter that exhibited almost no sensitivity.

p.8, It is noteworthy that RAC has chosen to explain that the research results of Los Alamos National Laboratory indicate that plutonium in the soil is insoluble. The interpretation should be that plutonium will, thus, not enter the ground water. RAC gives less commitment to that interpretation and states that plutonium "may not get into the groundwater." It is difficult to provide advice here except to note that it should be possible to incorporate a multiplicative parameter(s) to represent both the likelihood of water contamination as well as the degree. Maybe this has been done but it is not clear to me if it has.

Since we have committed only to completing screening calculations for the groundwater pathway, with the recommendation that future research be directed toward refining this calculation, we will not incorporate a calculation of this type. We will complete a calculation for the resident rancher scenario that includes the groundwater pathway, as well as one that incorporates only inhalation, with the understanding that the groundwater calculations are not definitive, but rather indicative of potential for dose.

Table 4. Soil-to-plant transfer factors should be noted to be chosen from NCRP 129 *recommendations*, not *data*.

We will note this in the text.

Units of pCi/g are used for the initial concentration in Table 4. Units of Bq/g should be used, though I am sure that RESRAD is probably to blame. In either case, it is inexcusable. Later on in the report (e.g., in Fig. 4), SI units are used. A consistent set of units throughout is preferable with SI being the preferred system.

Throughout this project, it has been difficult to stick to SI units, because the panel commonly prefers more recognizable units. We will insert both SI units and the readily recognizable conversion in all tables and within the text of this report.

The same comments as discussed in point number (9) above, apply to the parameters of "Plant food, contamination fraction" and "Drinking water, contamination fraction", both which are assigned a value of 1.0 in Table 4.

The value for plant food, contamination fraction was completely insensitive to change, so we left this parameter at its previous value. The drinking water pathway has been included only in a single scenario for the purposes of a screening calculation, and is intended to be conservative.

Groundwater/Drinking Water Pathway. It appears from this discussion that the parameter named "contamination fraction" refers to the fraction of the drinking water consumed that is contaminated. This is extremely vague. Does that imply that all water consumed is contaminated and only has a single concentration (that is, it never varies)? The assumption of 100% contamination with no assigned uncertainty is not credible.

Drinking water with a contamination fraction of 1.0 will come strictly from a groundwater source. The concentration will vary with the concentration of the ground water. As described above, we intend for any calculations that include groundwater as a source of drinking water or irrigation water to be conservative, bounding level calculations only, as a means of evaluating the potential for dose.

Furthermore, the chosen value of 2 L/d of contaminated drinking water is not realistic, but overly conservative. In regulating drinking water contaminants, EPA uses the value of 2 L/d for adults and 1 L/d for infants (10 kg body mass or less) as default values only. However, the most commonly cited study on water intake is that of Ershow and Cantor (1989, Total Water and Tapwater Intake in the United States: Population-Based Estimates of Quantiles and Sources, A report prepared under National Cancer Institute Order #263-MD-810264. Bethesda, MD: Federation of American Societies for Experimental Biology, Life Sciences Research Office) which estimated daily water intake and tapwater intake by age and gender. They defined "tapwater" as "all water from the household tap consumed directly as a beverage or used to prepare foods and beverages" and "total water" as tapwater plus "water intrinsic to foods and beverages". Values as great as 2 L/d can only apply to total water intake.

The all age-averaged median value for tapwater intake by males is about 1.1 L/d, and about 1.05 L/d for females. RAC should determine if gender and age-dependence will be accounted for. Regardless if age and gender-dependence is accounted for, realistic values for the population median tapwater intakes are only about one-half or less of RAC's presently assumed values.

Based on the above comments, the doses estimated in paragraph 4 are unrealistically too large.

Again, the use of drinking water in the soil action level analysis is done only to evaluate potential for dose of this pathway. It would not be possible, given the constraints of this project, to evaluate dose or soil action level including this pathway in any definitive way. We make these calculations and draw attention to this pathway only as a means of highlighting all of that which we do not understand.

P. 15. Change "daughters" to "progeny."

We agree and will make this change.

UNCERTAINTY DISTRIBUTIONS. The opening discussion of this section does not represent a state-of-the-art description of uncertainty analysis and the sources of uncertainty, as no distinction between uncertainty and variability is made and "uncertainty" and "variability" are sometimes misappropriately interchanged. RAC should be aware of IAEA Safety Series Report 100 (about 1990) or Hoffman and Hammonds (1994), "Propagation of uncertainty in risk assessments: the need to distinguish between uncertainty due to lack of knowledge and uncertainty due to variability", *Risk Analysis* 14, 707-712. This section should be rewritten to better distinguish uncertainty and variability.

At this place in our work, we do not plan to rewrite this section. The authors of this report have defined uncertainty within the context of the tasks we are accomplishing. Because of the implications of the results we will provide, we do not endeavor to quantify that uncertainty that results from lack of knowledge, particularly in the context of the groundwater pathway. We simply will not provide a set of soil action levels resulting from exposure to this pathway when so much about the transport within the saturated zone is not known. Our representation of uncertainty as encompassing variability is appropriate for this project and will be maintained.

Distribution Coefficient (p. 17): I adamantly disagree with the authors reference to "unquantifiable uncertainty." This is a prime example of the confusion between uncertainty and variability. For example, it may indeed be difficult to determine the extent of *variability* of this parameter (though there are numerous measurements reported in the literature). The uncertainty, however, can be estimated by the assessor (RAC in this case) based on whatever evidence and expert opinion they have. There is no single correct estimate of uncertainty as implied here, in other words, uncertainty is always quantifiable based on available evidence and judgment.

The discussion on the bottom of p. 18, which disregards certain data of Krey and Hardy (1970), Krey et al. (1977), is troubling. It is not possible for this reviewer to determine the legitimacy of RAC's analysis here. It is worth noting that Krey and Hardy had many years of study Rocky Flats contamination and they represented the finest sampling and environmental lab in this country, while the analysis of Rood (1999) is presumably a literature review. I recommend that RSALOP contract Krey as a reviewer of this material as well as of the Rood (1999) report. Krey is retired but can be contacted through the U.S. Department of Energy Environmental Measurement Laboratory in New York where he formerly served as Director.

RAC appreciates the comments of this reviewer, but we continue to assert our position about the degree to which the uncertainty about transport into groundwater is quantifiable. We appreciate and recognize all of the available data on transport into groundwater, but, for the benefit of the panel and this project, believe it is premature to evaluate uncertainty in this pathway and present a set of results that can be interpreted as applicable to the determination of soil action levels from this pathway. We would be remiss not to refer to the available research on the topic, but will not, at this time, present results with uncertainty bounds that have the opportunity to be misinterpreted outside the context of this study.

In the section on page 18, a discussion of K_d values takes place. We have obtained new data and are reviewing it for inclusion in this section.

P. 22, It is unclear what is meant by "RAC made adjustments to bring samples from various depths into conformity with the profile of Webb et al." Though it sounds like intentional manipulation of the data, it is probably more benign than that, but still not clearly explained.

What RAC has done is use the available concentration profiles reported by individual researchers and determine what the concentration in the top 3 cm was based on these profiles. In some cases, concentrations over depths larger or smaller than 3 were reported. In these cases, the fractional concentration depth profile provided by Webb et al. (1997) was used to adjust samples taken at different depths to a common depth of 3 cm. This is described in the text on page 20.

P. 22. It is unclear what RAC means that much of the data of Litaor could not be documented. I personally knew Mr. Litaor and he is an extremely thorough and careful researcher. Possibly the statement means that necessary ancillary data or sources of information was not provided. Mr. Litaor, however, can be contacted at his new employer in Israel for further information and I suggest that be pursued. His more recent publications in Health Physics give his present address.

We also have been in contact with Mr. Litaor throughout the course of this project. It is, in fact, because of Mr. Litaor's help that we were even able to obtain the database of values that he provided. We had some trouble, without constant contact with Mr. Litaor, discerning the depth to which some of the soil sample data provided were collected, because the references to the data were not readily attainable. Even after discussions with him, it was clear that the data provided to us were not separated within the database as to sampling depth. One set of data in which we were particularly interested was collected over "various depths up to 5 cm." The only option available to us was to assume the same depth of sampling (5 cm) for the data that we were not able to document. In this section, we simply warn the reader of the limitations of our data set. For the purpose of our spatial model, which is to provide a basis for integration of resuspension over large areas, the data set was sufficient. We continue to try to resolve these difficulties.

P. 24, The opening sentences describing a spatial model seem to me a bit elementary and imprecise. It would be better to describe that a spatial model is primarily intended to explain and/or predict the observations, thus allowing for predictions to be made at locations without observations with a reasonable level of confidence. Whether or not the model provides smoothing is entirely optional. While most do, I certainly don't agree with the statement that it *must* do so.

The uniformed reader might be led to assume that the two methods (kriging as used by Litaor) and determination of power functions within polar sectors (as used by RAC) are equal. They are not, as their origins and technical basis are so different, it is difficult to compare. Kriging intentionally takes advantage of the spatial correlation of data and uses that to an advantage when predicting values at locations where no observations are available. RAC notes that in two sectors (292.5° and 315 °), there was too little data to determine the functions, thus RAC assumed the functions from a nearby location (270°). It is worth pointing out that kriging would base these locations on the spatial trends, rather than on an assumption. I am not suggesting that RAC revise their methods of spatial interpolation to kriging (which is a much

294

more difficult mathematical technique) but am pointing out that it could be of some advantage, such as in the situation noted here.

We intentionally selected the power function analysis to base our contour smoothing on the assumption that the spatial signal was the result of wind transport of contaminated soil particles from the 903 Area. A kriging analysis was not justified in the context of what we were trying to accomplish.

Fig. 4. Along a west-east line at coordinate of Northing 441.0, there is a line of measurements that are all gray circles (10-100 Bq kg^{-1}), yet they fall well outside the 2 Bq kg^{-1} contour. Where is the discussion explaining these measurements and the lack of agreement of the contours with the measurement data?

As with any model, the model described here is not capable of predicting every measurement. Because our model based the spread of contamination on the assumption that wind transport was responsible for the spread of contamination, there are measurements outside of our wind transport contours that likely resulted from other contamination events at Rocky Flats. Evidence from the dose reconstruction studies as Rocky Flats might give us some insight into the source of these above background readings. A fire at building 771 at the Rocky Flats plant in 1957 released a significant amount of airborne particulate contamination. Meteorological data from that period indicate that the wind direction probably directed the contaminant plume in a southerly direction before the wind direction shifted and the plume proceeded to the northeast. Although particle size of contaminants was very small and little deposition probably occurred in the aftermath of this event, it is likely that the measurements taken at these locations resulted from contamination from the 1957 fire.

p. 28, RAC states they "will estimate the variation of the air concentration that exists within the defined domain based on the current state of ground cover, using the existing air concentration data." I have two questions about this statement. 1) The air concentration data can obviously be used to estimate its own variation. Is there something more important being said here? 2) RAC has claimed in the past the importance of using a dynamic model (which implies incorporating a time-dependence to estimate values likely in the future). How will the current state of ground cover be extrapolated to the future for the purposes of dynamic modeling?

The current state of ground cover gives us an important stepping off point. To this estimate of dust loading determined using the available soil and air concentration data, we can apply an enhancement factor that uses the resuspension studies completed at Rocky Flats to estimate the increase in dust loading that might result from an event that would remove the available vegetation cover.

p.28. The paragraph beginning "A procedure such as this..." needs rewording. Obviously some words are left out which render the paragraph unintelligible.

It appears a word was left out of this paragraph during review. We thank this reviewer for bringing it to our attention.

p. 28-29. The discussion on the fetch of airborne dust incorporate opinions of RAC ("these distances seem to short to be consistent...") and the validity of those opinions versus the findings in the literature is a very technical matter. I suggest the RSALOP contact Dr. Joseph Shinn to evaluate this discussion. It is important and deserves an opinion of greater expertise than my own or anyone on the RAC team. Shinn can be contacted at Lawrence Livermore National Laboratory in Livermore, CA.

p. 29-30. The discussion of mean annual wind speed seem reasonable to me though the findings are outside of my expertise. The opinion of Dr. Shinn would also be valuable here.

While this project would certainly benefit from a rigorous review process including reviewers from a number of arenas, there is a time limitation that will prohibit additional review at this stage of the project.

P. 32. The full-time resident rancher is an unrealistic scenario, being that the assumption is that members of the family never leave the site. As a reviewer, possibly I have not been given an adequate briefing on how the scenarios are to be defined and used, but such assumptions are not realistic and contribute little to an understanding of the risks of RFETS. I recommend changing all unrealistic assumptions because they have no face validity and no place in the application of probabilistic risk assessment. Such scenarios do not require peer review because they have no basis on which a review can be conducted. I do not endorse these values or any unrealistic scenarios.

It is important to understand the context of the development of the scenarios, which were carefully established with the help and consensus of the panel. The process by which these scenarios were developed was long and involved. That process can not be fully outlined here, but suffice it to say that the scenarios have been carefully thought out by both RAC and the panel, and represent our collective view of reasonable scenarios for a future that is impossible to predict.

Table 10 is a summary of parameter values, most of which have been commented on above.

The number of days per year in which soil ingestion is assumed to take place is excessive. Northern Colorado where RFETS is located, normally experiences cold weather that would make it impossible for a child or infant to have access to soil every day. Protection of the public can be adequately ensured by setting the upper end of the distribution equal to 365 days, not the median.

Again, the panel has decided upon this value, which is a constant.

P.34. The first paragraph on this page explains the review of literature data, defining distributions, etc. The 2nd paragraph attempts to explain, but actually glosses over without adequate explanation, a very important concept. Here it is described how a percentile is selected and the rest of the data disregarded. It appears that a single value of each parameter is chosen which RAC believes is protective of the population and the entire set of single values (one for each parameter) are then used to calculate the soil action level (I assume). The question is: How reliable of an estimate is produced? It has long been known that choosing conservative values for

296

all parameters results in a highly exaggerated final result. Possibly I have missed something, but I don't understand this process and I express great concern over what is written here.

This technique was discussed and agreed upon by the panel.

I note further that the last sentence of the report (p.41, "Values for the soil action level and dose will be presented as distributions of possible values for each individual scenario") seems not to be in agreement with the process of fixing values as described on p.34.

We intend to fix values only for the scenarios (Table 10), allowing the parameter values that fall outside of the boundaries of the scenarios (Table 4) to vary. This will provide a distribution of doses and soil action levels for each fixed scenario.

Breathing rate, 2nd paragraph (p.34). The word activities is overused in this sentence ("...the activity levels for indoor and outdoor activities differed").

We will adjust our word use in this sentence.

Groundwater (p. 40). RAC has chosen to evaluate contaminated groundwater as a source of exposure and this seems like a reasonable thing to do. RAC should be cautioned, however, that their last statement ("Failure to address these pathways quantitatively leaves open the question of their potential importance") implies that they are interested in correctly *quantifying* the risk. For that reason, they should use all of the quantitative evidence, including the insoluble nature of plutonium as assessed by Los Alamos National Laboratory. Ignoring any evidence will defeat the process of correctly quantifying the risk.

We will change the wording in this sentence to reflect our intent to provide a screening level calculation, not a quantitative risk evaluation, for the groundwater pathway.

Drinking Water Intake (p.40). I have already addressed the overestimate of water intake that RAC proposes. Does Layton (1993) really address water intake? I only remember that it discusses inhalation rates.

Thanks to the reviewer for noting that the reference was not the appropriate one. The correct reference is *Finley et al. 1994*, and the change will be made in the final report.

Reviewer C

Introductory Note: for convenience, overall comments are presented first, and more detailed comments are presented on a page-by-page basis. Purely editorial comments are introduced by the word "Editorial". From my perspective, RAC need not respond in writing to any of the comments and suggestions labeled "Editorial".

Overall Comments

This is a well-conceived, well-presented and well-written draft, and was a pleasure to read. It is important that this task is in very good shape at this stage, since arguably it is one of the most important in the whole project. There were very few typographical errors, and only very few sections merit substantial re-writing or additional content for improved clarity and comprehension.

The Executive Summary was particularly excellent. Anyone who reads and fully understands the Executive Summary has a very good understanding of the entire report.

I recommend that a paragraph providing an overall perspective be added to the Most Sensitive Parameters section in the Executive Summary. It should provide RAC's general view on the reasons it has chosen different values for the five parameters, such as: RAC is using more recent or more extensive data, DOE/EPA/CDPHE did the best they could at the time, DOE/EPA/CDPHE really chose poorly for some of these five parameters, DOE/EPA/CDPHE badly botched the job back in 1996, etc. This perspective will be very important for the non-specialist reader who reads only the Executive Summary of the results of this task. If such a perspective is not provided, it will leave each reader free to draw his or her own conclusion from among the choices I listed. As an example, later on page vi, RAC clearly points out that for the soil-to-plant factor, RAC used a more recent definitive report, which was simply unavailable in 1996. This choice would be understandable to and accepted without question by all but the most cynical and suspicious readers, and should be part of the overall perspective that I recommend be added to the Executive Summary.

A very good suggestion for improvement of the executive summary, which we intend to take.

Detailed Comments

Page v, end of second paragraph. Either here or somewhere in the Executive Summary there should be a brief description of: a) the major conceptual difference(s) between RESRAD 5.82 and Version 5.61 used in 1996 and/or b) the major differences between the two versions as they relate to this specific project. See, for example, page 2, 2nd paragraph, where this is dealt with.

We will incorporate this enhancement

Page v, last paragraph. This paragraph, which introduces RAC's "bottom line" values as shown in Table ES-1, should be expanded to provide a little more explanation of how RAC

298

reached its values, or else it should alert the reader that the reasons for any differences will be explained in detail later.

As a part of the general comment above, this will add to the clarity of the executive summary, and we will incorporate a discussion like this.

Page vi, first paragraph, next to last sentence. There should be a brief description (a phrase would do) explaining why RAC's value for uranium is four times higher.

We continue to explore the topic of K_d values, using more recent data from the actinide migration panel. We will present final values in the final report.

Editorial, page vi. Is there a need for a brief description, perhaps in a footnote, about the use of the geometric standard deviation, and why this rather than some other statistical measure of variability was chosen by RAC?

In general, the distributions were either described in the literature as lognormal or the distributions created from the available data fit best to a lognormal; the statistical measures selected to best describe this distribution were the geometric mean (median) and geometric standard deviation.

Page vii, first and second full paragraphs. I strongly endorse RAC's approach to use actual air and wind data. In particular, if there is any suggestion that RAC should revert to the 1996 value for mass loading, I urge that RAC hold firm in its choice.

We plan to stick to this approach.

Editorial. Page 1, 2nd and 3rd paragraph. Some language should be added to distinguish the Monte Carlo feature in the new version of RESRAD from the Monte Carlo interface developed by RAC, just to avoid confusing non-specialist readers.

We will incorporate this enhancement.

Editorial. Page 1, 4th paragraph, 4th sentence. Can some qualifier be put on "large", say, XX% change? Alternately, could there be a definition in the next sentence, where sensitive, limited sensitivity and no sensitivity are listed?

We have incorporated qualifiers into these sections of the report.

Editorial, and perhaps more than that. Pages 2-3, Differences between... This section (especially the first paragraph) needs some clarification and elaboration, if for no other reason than the roughly 5-6 fold increases in the soil action levels for plutonium shown in Table 1, which leap out at the reader. First, aren't there *two* changes (not one) between the two versions, the change in the air concentration and the addition of wind speed? In the text, can you provide some perspective on the relative importance of the two? Also, is "adjusted" a better choice than "altered"? Should "overly" be inserted before "conservative"? I suggest that RAC take a fresh look at this entire section with the goal of making it more explicit.

This section of the report, in particular, has spurred a great deal of discussion and even controversy. A comparison of the two versions of RESRAD used during this study was included in this report only as a means of illustration. We intended to show that the resuspension mechanism (the single change we refer to) in the more recent version is significantly different than in the previous version. What we propose because of this difference is a resuspension calculation based on actual site measurements as opposed to this generic, and generally unsatisfactory when viewed from an output perspective, resuspension calculation. We plan to clarify and make explicit the point of this section in the final version of this report.

Editorial, page 3. In my copy of the draft, there is a speck of black that on first reading turned 1088 into 1.088. I trust it was added by the copy machine, and does not exist on the original.

The copy machine did add the speck of black; the value on page 3 should read 1088, not 1.088.

Editorial. Page 5, 1st paragraph, last sentence. Substitute "believe" for "feel".

We will make this adjustment.

Editorial. Page 5, 2nd paragraph. RAC selected 0.03 meters to maintain consistency with *which* definition, the one for soil mixing layer or thickness of the contaminated zone? And why is RAC comfortable being consistent with inconsistent definitions? Is the phrase "surface or resuspendible soils" the best one available?

As mentioned in the response to comments from Reviewer A, it is likely that the discussion in this section is not adequate to describe what we intended with the selection of 3 cm as the depth of resuspendible soil. We will adjust this discussion to be more consistent with our intent, as described in the response to Reviewer A.

Editorial. Page 5, 2nd paragraph and 5th paragraph. Perhaps there should be a little more explanation of the use of 0.03 meters for depth of mixing layer versus 0.2 meters for thickness of the contaminated zone.

As a result of the significant number of comments about the above two quantities, we will look at reworking the section which explains the use of the two values.

Editorial. Page 5, 3rd paragraph. Would RAC be comfortable adding "very" before "conservative" in the last sentence of this paragraph?

300

Several reviewers had a comment about this quantity for indoor dust filtration. This was the quantity used in the previous analysis, and RAC saw no reason to change the value for the present analysis. We plan to explore the use of a distribution for this value.

Editorial, page 7, 3rd-4th lines. The exponent got bumped down a line.

We will fix this for the final version.

Editorial. Page 9, Table 3 and following paragraph. "(DCF)" should be added to the heading on the table, and "f_1" is not defined either in a footnote to the table or in the text.

We will make these changes.

Editorial. Page 11. I recommend that the order of the parameters and the two columns be identical to those in Table ES-1 on page vi. Also, would it help to break this mega-table into a set of tables? In particular, for the parameters not exhibiting sensitivity, should there be one table for the ones where DOE and RAC values are different, and a second one where they are identical? Finally, shouldn't "not" be capitalized in the heading of the last group of parameters?

We will put the parameters in the same order in the two tables. We have struggled with the readability of this table, and will continue to make adjustments to make the table easier to read.

Editorial, and perhaps a bit more. Page 13, first paragraph. I suggest language be added explaining the utility of including the "bounding level, screening calculation" for the one scenario, including stating whether it is meant to provide an upper bound or conservative estimate.

The bounding level, screening calculation is important primarily for the sake of completeness in the review. We recognize that we cannot make a detailed quantitative evaluation of the dose from the groundwater pathway, but we would like to provide perspective and perhaps some encouragement to explore future work in this area.

Page 15. If possible, could RAC be a little more descriptive of the type of study it believes necessary, and in addition, provide recommendations on how RAC's own final results (whatever they may be) should be re-visited when such work by others is complete? This might include running sensitivity studies, for example.

The Task 5 report might be a better place to provide recommendations on how our final results might be revisited at a later date. Such recommendations will be incorporated in that report.

Editorial. Page 17, 3rd full paragraph, last sentence. Substitute "The" for "A".

We will make this change.

Editorial. Page 20, 2nd and 3rd paragraphs. Did RAC "define" the model, or did RAC "build", "develop", or "construct" the model?

We will include more appropriate wording in these locations

Editorial. Page 22, 2nd complete paragraph, and page 23, legend for Figure 2. Was Litaor's contribution in this regard so great that it justifies a complete name, the only individual so honored in the entire report? Also, more facetiously, does the name Iggy generate a high degree of technical confidence in the average reader? (Even RAC rejects much of Iggy's data in the 3rd paragraph on page 22.) I suggest either just using the last name, or M. I. Litaor.

We will use M.I. Litaor to refer to this individual the first time.

Editorial. Page 29, 2nd full paragraph. Insert "and" before "annual" in the next to last sentence.

We will incorporate this change.

Editorial, Page 31 ff. Using "current" to describe the 1996 scenarios bothers me somewhat, especially since "current" is also used to qualify the onsite worker scenario. Labeling them as "1996 scenarios" also doesn't seem quite proper, though strictly speaking it would be correct to do so. Since RAC has four and the 1996 effort had three, perhaps the editorial solution is to describe the origin of the three in one place, as in the 4th paragraph on page 31, and then later identify them as the "three scenarios" or "the DOE/EPA/CDPHE scenarios" both in the text and in tables (such as Table 10). RAC's can be identified as the "four scenarios" or the "RAC scenarios", as appropriate.

We appreciate this comment, and will do everything we can to clarify the language within the report, making it clear at all times to which project and which scenarios we are referring.

Editorial. Page 40, 3rd complete paragraph. The exponent on "d" should be –1, as it should also be for "y".

We will make this change.

302

Reviewer D

This is a well-conceived and useful draft report, as was the Task 2 Report by the same authors. Prior to commenting on that earlier report, this reviewer raised a number of concerns regarding the assumptions underlying the DOE/EPA/CDPHE application of EPA's 15/85 mrem/y dose criteria and their choice of exposure scenarios for implementing those criteria via soil action levels, including the selection of parameters characterizing the individuals exposed. This Task 3 draft begins to address many of those concerns. At the risk of boring the reader of these comments, and since paper is cheap, I repeat here the basis for those concerns before commenting on how this report addresses them:

1. Misuse of EPA's draft 85 mrem/y criterion.

This criterion was proposed to assure protection during unanticipated failure of institutional controls only. It was not meant for planned land uses in the distant future when controls are assumed to no longer exist. EPA requires review of institutional controls no less often than every five years as long as they are needed to meet 15 mrem/y (40 CFR Part 300.430(f)(4)(ii)). Failures are expected to be of short duration and corrected when identified. In EPA's current regulations (OSWER Directive No. 9200.4-18; August 1997) the 85 mrem/y criterion has been dropped – it is assumed unnecessary under the periodic review requirement.

It appears that reasonable assurance of effective long-term institutional control at Rocky flats for the duration of the hazard is not now available and is, in fact, probably not possible. Accordingly, cleanup of the entire site to 15 mrem/y now, without reliance on controls, is, realistically, likely to be needed. The choice of exposure scenarios for the Tier I Action Levels for the so-called "buffer" and "industrial" areas is affected, as well as for areas *outside* the buffer areas, since these locations clearly must meet 15 mrem/y under unrestricted use in any case, and the action levels for the immediately adjacent buffer area, under the existing proposal, permit significantly higher levels.

We will be completing calculations using both the 15 and 85 mrem y^{-1} criteria, presenting these values to the panel, and allowing them to make recommendations based on these results. The panel could likely use this reviewer's comments to expand its understanding of this topic.

The draft report proposes two new exposure scenarios that go a long way toward providing the basis for satisfying the above needs: the "current site industrial worker" and the "resident rancher." With respect to the industrial worker scenario, I assume that the choice of 60% of time spent outdoors reflects the seasonal nature of outside work and that this scenario could therefore reflect a grounds maintenance worker. However, the assumption of no onsite drinking water does not appear justified for such an individual.

The groundwater/drinking water calculations will be completed only for the residential rancher as a bounding level, screening calculation to provide some perspective on the potential for dose from this pathway.

There are more serious problems with the resident rancher scenario. I assume that it was considered more reasonable to posit a resident rancher than a rural resident based on current land uses (no explanation is given in the report). However, given the present the rate of expansion of populations in the Denver area and the extremely long duration of this hazard, that choice would appear to be extremely difficult to justify over the long term, and no justification is provided in this report. It also is not clear what the justification is for selecting only 40% time outdoors for a resident rancher (rather than 60%, as in the case of an industrial worker), nor is it clear why this scenario is restricted to locations east of the 903 area (instead of including that area). The report needs to modify these assumptions or provide a convincing rational in support of them. (See also the comments below on the definition of the RME individual required to be protected under CERCLA. It would not take many rural residents to constitute their designation as the RME individual.) Comments on the usefulness of the infant and child scenarios are provided later. Incidently, the headings "nonrestrictive" and "restrictive" appear to be reversed in Table 10.

Thanks to the reviewer for noting the reversal of the heading in Table 10; this will be changed in the final report. These scenarios were selected after many long discussions with the panel and were approved by them in May. The scenarios were designed to address not only what we know about the possible future at Rocky Flats, but also what we do not and can never know about events that have not occurred yet. We will elaborate on our discussion on time indoors and outdoors for the scenarios in the final report.

2. Inadequate Exposure Scenarios:

My previous comments on this topic were: "Under CERCLA, the choice of exposure scenarios is intended to assure protection of the "*Reasonably Maximum Exposed*" (RME) individual. This is not the same as the *average* member of the affected population, nor is it the *most* exposed individual. EPA has devoted considerable effort to clarifying this admittedly elusive concept. The following quotes are typical of EPA 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... The intent of the RME is to estimate a conservative exposure case (i.e., well above the average) that is still within the range of possible exposures.*' ("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 90^{th} percentile of the actual (either measured or estimated) distribution. The conceptual range is not meant to precisely define the limits of this descriptor, but should be used by the assessor as a target range for characterizing "high-end" risk.*' ("Guidance on Risk Characterization for Risk Managers and Risk Assessors," Memo from F. Henry Habicht II, Deputy Administrator, EPA, to Assistant Administrators and Regional Administrators, February 26, 1992.

304

"A number of the choices in the DOE report do not appear to meet these criteria, but instead are more reflective of average populations or behavior of individuals. For example, the office worker chosen for the industrial area scenario reflects the average worker for the assumed use of the area as office buildings. However, an RME individual at such a site would be a maintenance worker, who takes care of the assumed "well-maintained landscaping" (DOE report, p. 6-16). It is also not at all clear that the "industrial" area would be exclusively used, for office buildings for the duration of the hazard. Given the relatively remote location of the Rocky Flats site, it appears optimistic to assume that use of this area would be so limited. A more realistic scenario would envision more traditional industrial uses, such as lumber yards, light industry, or even scrap yards. Under these uses the office worker scenario becomes untenable as the basis for deriving soil action levels.

"A similar difficulty arises for some of the choices of exposure parameters for the individual scenarios. For example, the exclusion of ground and surface water in the rural residential scenario does not appear to reflect the RME individual. What assurance is there that less than 10% of individuals would not avail themselves of existing ground or surface water at any point in time during the next 1000 years? The existing ground water appears adequate for subsistence living, and quite adequate for use for limited irrigation, as for a family garden. If non-use of ground or surface water is an assumption, rather than a condition assured through an institutional control, it is not an appropriate element of the exposure scenario. (In any case, in the scenario for the 85 mrem/y criterion, when institutional control is assumed to be absent, non-use clearly should not be assumed.). Other parameters that warrant examination are the assumption of no contamination, now or in the future, below 15 cm, when plant roots are assumed to penetrate to 90 cm, and the degree of retention of mass loading for foliage assumed for this semi-arid area in the rural residential scenario; the assumption of no use of surface or ground water and the time limitations on annual usage by the RME individual in the buffer zones; and, for all the scenarios, the blanket assumption of Class Y solubility for plutonium under all pathway conditions."

The present report addresses many of these problems. Importantly, in addition to the new scenarios noted above, it adopts the 95% breathing rates, in conformance with the RME individual, and includes (to the extent feasible) bounding doses due to ground water for a number of the scenarios. There are, however, some remaining problems.

The report adopts, as plausible, all three scenarios outlined in the Rocky Flats Cleanup Agreement. This is not reasonable, since these do not satisfy the CERCLA criteria outlined above: The "office worker" is clearly not an RME individual; ground water intake is still not considered for the "resident;" and, at least according to Table 10, both the "resident" and the "open space user" spend 100% of their time indoors!

We have accepted the DOE scenarios as part of the total scenario analysis for this project. The results of the calculations will be provided to the panel, and the panel will have a chance to make recommendations based on the results of all of the scenario calculations.

It is important to point out that although the resident and open space user spend their time indoors, the air concentration indoors has been set to be equal to the air concentration outdoors.

The infant and child scenarios omit onsite drinking water (infant formula is made with water, and children drink the same water as their parents!).

The drinking water analysis will only be presented as a bounding level, screening calculation to present the potential for dose from this pathway as a means of encouraging further research in this area.

The report omits irrigation water from the child scenario, but include it for their parents (children eat most of the same produce that their parents eat).

Again, this calculation is not meant to be quantitative in terms of providing boundaries for dose, but rather to present screening results.

Finally, I have a major conceptual problem with the use of infant and child based scenarios. They misuse the annual dose criterion by artificially inflating its effect. The basis for the dose limit is lifetime risk, which already includes the risks due to exposure during infancy and childhood. The annual dose criterion is a useful surrogate for lifetime risk only if it is applied to standard man, and was never intended to limit annual risk to a uniform value for any age individual. (If that were true, permissible annual doses for senior citizens would be extremely large.) I strongly recommend dropping these scenarios.

These scenarios are very important to the panel, as a means of presenting results that are meaningful to all possible recipients of dose. For parents living in the vicinity of the plant, this means that their children need to be assured of protection. We will take this reviewer's comments to heart in our presentation of the results for these scenarios.

Other Comments on the Task 3 Report.

The report should at least comment on the subject of co-variance, in the context of the use of single-parameter analysis (p. v).

Co-variance suggests the possible correlation of parameters. Although a single-parameter analysis ignores possibilities of correlation, there is some possibility of this, which we did consider while completing the analysis. We will consider adding some text that relates to this.

It is not clear that the use of existing actual air monitoring data can approximate future land use conditions that do not now exist - e.g., agricultural use under drought conditions (witness current mid-Atlantic agricultural regions). I suspect that such a procedure would underestimate inhalation doses due to resuspension (p. vii). In this regard (the degree of conservatism appropriate), to what extent can we predict the effects of climate over a 1000-year period on enhancement of resuspension?

We intend to present enhancement factors that simulate these types of conditions and make the resuspension pathway more broadly applicable to the range of possible future conditions at Rocky Flats.

Endorse the proposed use of current estimates of fruits, vegetables, and grains, especially in view of current dietary trends (p. ix).

The choices for parameters with limited sensitivity appear logical (pp. 4-5).

The discussion of the gamma shielding factor represents an improvement (p. 6).

The treatment of ground water ingestion (pp. 13-15) confirms that more work is needed on this potentially important pathway, especially with respect to colloidal transport of americium. The observation of Honeyman that study conditions (increased well pumping) in the Kersting et al. work may have enhanced colliodal concentrations is provocative – that is just what extensive ground water use would do.

We thank this reviewer for all of the above comments.

Figure 4 suggests that some supplementing of the spatial soil model may be desirable to accommodate the higher measured values at the bottom of the figure, which appear to be an order of magnitude higher than the model predicts.

We continue to review the spatial soil model for improvements through the production of the Task 5 report.

Would it make sense to use the 95% value for soil ingestion, but multiply it by seasonal and weather-based soil availability factors (e.g., 0.5 for frozen or snow covered, and 0.7 for rainy weather during the balance of the year, or 0.73x0.5x0.7=0.26)?

This parameter has been extensively discussed by the panel, and the values presented represent the final conclusions regarding this parameter.

Minor comments.

1. Is there any way to provide for the possibility of colloidal transport in the uncertainty analysis?

Not within the boundaries of our screening only analysis.

2. Has retention of foliar deposition been evaluated for the semi-arid conditions at Rocky Flats?

We will look into availability of data of this type.

Reviewer E

General Comments

1. The report lacks a complete overview of the sensitivity analysis performed. The following two questions are left unanswered.

a. Why was the sensitivity analysis limited to site-related parameters? For the convenience of the reader, the universe of input parameters to RESRAD should be categorized and it should be clearly stated in the introduction and executive summary which categories of parameters were evaluated in the sensitivity analysis, which were not, and why. For example, two obvious categories are:

site -related (or environmental fate and transport) parameters (e.g., those listed in Table 4), and

exposure-related (or scenario) parameters (e.g., those listed in Table 10).

The sensitivity analysis was limited to the site-related parameter because only these parameters will be treated stochastically in the soil action level analysis.

Although the RFP and RAC's proposal did not limit the sensitivity analysis to site-related parameters, that is what apparently was done. There may be good reasons for this. They should be made explicit.

The sensitivity analysis was limited to site-related parameters as agreed upon by the panel. Scenario-related parameters represent human characteristics or habits. For our hypothetical individuals, we assume that we understand the characteristics of a specific individual, but present a variety of scenarios so that many different types of individuals are represented.

b. Which exposure scenarios were evaluated in the sensitivity analysis? If all scenarios were evaluated, were the results consistent for all (i.e. were the same parameters sensitive for all scenarios? (For example, p. vi, par. 1 implies that Kd was only sensitive for the rancher scenario where groundwater was considered as a source of drinking water. Is this the case or was Kd important for all scenarios?). It seems that there would be a way to create a table illustrating (qualitatively or quantitatively) which parameters were important for which scenarios to provide a summary answer this question.

No exposure scenarios were evaluated in the sensitivity analysis. Each scenario represents a single individual with unique physical and behavioral characteristics. These characteristics include variables correlated with dose, such as average breathing rate or dietary habits. As explained in the report, we used a wide range of references for information on these parameters. Then we generated a distribution of values and sampled from the distribution, using Monte Carlo techniques. This process considered the available studies equally. We selected a certain percentile from that distribution for each scenario. Once a parameter value was selected from our distribution of values for use in the scenario, the scenarios were considered fixed.

308

2. The ultimate purpose of the current analysis, as I understand it is to develop revised soil action levels for RFETS, where, using RAC's words, a radionuclide "soil action level is a concentration of radionuclide in the soil established to protect people from receiving radiation above a set limit "(p.v). I assume radionuclide soil action levels (RSALS) will be used as soil remediation goals at RFETS. Yet, it seems that RAC has focussed a lot of effort on setting up a *baseline* risk assessment by developing contours of actual contamination levels to specify initial contamination concentrations and areas for use in RESRAD and using site data to develop relationships between contaminant concentrations in air and soil for use in the resuspension calculations. I agree that this approach will make, as RAC states "the calculation of dose more meaningful"(p.viii).

However, it is the dose due to due to current contamination levels that will be calculated. I'll call it the baseline dose, here. I think that RAC's proposed analysis makes the baseline dose more meaningful, but is not feasible for calculating RSALS. To develop RSALs, one needs a different analysis, the purpose of which is to assure that the dose at the RSAL (or post-remediation radionuclide concentration) is less than or equal to the target dose with some level of confidence.

I have some questions about whether RAC's approach outlined in Task 3 will lead to meaningful RSALs in Task 5. RAC makes the claim that their procedure to calculate resuspension parameters (described under the heading "Mass Loading Factor" p. 27) will be used to " estimate annual average plutonium air concentration at any location at or near the site"(p.28, par.. 4) They go on to say that they "*may* [emphasis added] also estimate plutonium air concentrations based on the assumption of reduced soil concentrations that simulate the results of remediation" (p. 28, par. 3). Isn't the latter the point of the whole analysis--which is to develop RSALS? Additionally, even if the relationship between current soil and air concentrations is elucidated for the baseline risk assessment, what assurance is there that the same realtionship would be appropriate for a remediated site?

RAC justifies their approach for calculating the resuspension parameters based on the fact that " air concentrations in the domain of a scenario depend not only on soil contamination within that domain, but also on soil contamination throughout a larger region" (p.28 par. 4). I do not question that this is an important consideration in a baseline risk assessment. However, I wonder how this can be accounted for in the development of RSALs since you would not know before a remediation effort exactly what the contaminant concentrations in soil would be following the remediation effort . It seems to me that at best you have to assume that the entire area is uniformly contaminated at the RSAL (since theoretically that would be the goal of the remediation effort). I suggest that the original approach in the DOE/EPA/CDPHE (1996) analysis for setting RSALs where it was assumed that there is a large area with uniform contaminant concentration.

The bottom line is this: It seems to me that different methodologies and inputs are called for in calculating baseline risk and RSALs. I think RAC needs to be very clear about the methodologies and inputs they are using for each. In addition, the panel needs to be clear about which analyses it wants RAC to perform.

While this reviewer may not understand the fundamentals of the approach we are taking here, we want to assure the reviewer and the panel that this analysis will produce the desired results, as will be shown in Task 5.

Specific page-by-page comments:

1. Contents. I suggest some modified headings that reflect my general comment no. 1. 'SENSITIVITY ANALYSIS' and 'UNCERTAINTY DISTRIBUTIONS' should be secondary to a heading like 'SITE-RELATED PARAMETERS'. Similarly, 'SCENARIOS' should be renamed to something like 'SCENARIO-RELATED PARAMETERS' (this section should include a brief introductory statement that points how that scenaio-related parameters will be treated deterministically in the analysis).

At the very least, we will include a statement about how scenario-related parameters were treated deterministically.

2. p.v , last par.. Suggested revision for second sentence which as it reads now appears to confuse uncertainty and variability.
" The probability distribution functions describe the uncertainty in the parameter values that arises due to ....."

We will carefully consider this suggestion and look at revising this sentence.

3. p.v. Regarding the use of the term 'distribution coefficient'. At least at first --in the exec summary and intro-- be more specific. Replace with 'soil-water equilibrium distribution coefficient'. In general in environmental fate and transport modeling, there are other types of distribution coefficients.

Good suggestion – we will make this adjustment.

4. p.vii. par. 1. Start with "The term 'mass loading' is used in this analysis as..." Here, too, there is no standard definition for 'mass loading' in environmental fate and transport modeling. To avoid confusion, just be clear about your definition for use in this analysis.

We will make this adjustment.

5.. p. vii, last par.. Bullet the list of five less sensitive parameters to make it easier on the reader.

We will make this change.

6. p. ix. before last par.. Make it clear that deterministic values will be used for scenario-related parameters in the assessment.

We will make this clear in the final report.

310

7. p.1, par. 3. It is not clear at this point (and it should be) why RAC has developed a Monte Carlo interface for RESRAD when in the previous paragraph it says RESRAD has one already.

The interface built into RESRAD that was used in the sensitivity analysis was built on Monte Carlo principles, but accomplishes only a single-parameter sensitivity analysis. There is an additional interface built in to RESRAD that supposedly creates uncertainty distributions, but which the authors of this report had no luck getting to run. Nonetheless, it is important for *RAC* to develop their own Monte Carlo analysis for two reasons. 1) It is a contract requirement that we build a Monte Carlo interface, and 2) We needed to build our own module to incorporate the alternate calculation of resuspension.

8. p. 4 par. 3. It seems more appropriate to have performed the sensitivity analysis using the total possible range of values for all the parameters rather than to have varied the parameters by a factor of 10 in either direction.

The analysis could have proceeded in many different directions, but we chose one and stuck to it.

9. p. 9 1st par. under 'Remaining parameters', 1st bullet. Isn't K_d a saturated zone parameter? Perhaps this bullet item needs to be more specific or needs to specifically exclude K_d.

We will make a change that will exclude K_d from this list.

9. p. 11 Table 4. Most, but not all of the information from Table ES-1 is repeated here under 'sensitive parameters'. Table 4 should be at least as complete as Table ES-1 or it should just refer to Table ES-1.

We have had another comment on this, and will make the appropriate changes.

10. p.18, 2nd par. under Table 6, last sentence. Be more specific about what you mean by the 'midpoints of the Kd values from Table 5'.

Thank you for this comment. We will attempt to be clearer in the final version of the report.

11. p. 20 par. 2. This paragraph starts with "To resolve this problem...". It is unclear how this resolves the problem.

We solve the problem presented by RESRAD (homogeneity of contamination required) by incorporating our own spatial soil model that allows heterogeneity of soil contamination to exist.

12.. p. 31 2nd par., last sentence. Makes no sense. Re-read. Re-word.

We will work to clarify our view of scenarios.

13. p.31 4th par., last sentence. Redundant with 1st sentence.

We have had several comments of this sentence from good, careful reviewers. We thank this reviewer for this comment and will change this sentence.

14. p. 33 Table 10. 'Soil Ingestion' in first column should be in units of g/d. Otherwise it looks like 0.2 g/ 365d which is 0.0005 g/d. With this change, might have to clarify the wording under the open space scenario.

We will work to make this section of the table more readily understandable.

15. p.33 Table 10. Why is there 'NA' entered for drinking water ingestion under the infant of rancher and child of rancher. If the adult rancher drinks the well water, why don't the infant and child?

We are conducting a groundwater/drinking water analysis only as a means of presenting the results of screening calculations. We have agreed to include the pathway for only one scenario, the residential rancher.

16. p.33 Table 10. Why is there 'NA' entered for the 'Fruits. vegetables and grain consumption' of the 'Infant of rancher'. p. 40 indicates that this value should be entered as 200.

16. p.33 Table 10. Why is there 'NA' entered for the 'Leafy vegetables' of the 'Infant of rancher'. p. 40 indicates that this value should be entered as 26.

There appears to be a typo. We will make the table and the text consistent.

17. p.37 1st par.. second to last sentence. Give the units on the 'geometric mean of 0.2'.

Thank you – we will make this change.

References

US DOE, US EPA, CDPHE (1996) Action Levels for Radionuclides in Soils for the Rocky Flats Cleanup Agreement Final.(October 31, 1996).

312

PANEL COMMENTS

Victor Holm

I was impressed with your Task 3 report. First it was well organized and very readable. Your early decision to concentrate on a few parameters that are most sensitive has served to focus attention and prevent endless debate over matters that have little or no practical value. I was especially impressed by the way you integrated the many previous studies at Rocky Flats into the work, especially the sections on Area of the Contaminated Zone on pages 19-27 and the discussion of the Distribution Coefficient. Sometime important data affecting a parameter are discussed in a different section, but short of repeating the data in both sections, I don't see a solution.

The discussion of scenarios seems to fit better here than in Task 2 and the discussion is much more complete than in the draft.

I, along with Bob Kanick were instrumental in selecting a quantitative risk assessment for this study. The reason for this was the expressed concern by several members of the panel that safety factors be incorporated in the final result. We understood that if safety factors were incorporated individually in each parameter there would be no way to evaluate what the final safety factor might be. Secondly many of the parameters did not lend themselves easily to quantified safety factors. Instead what we hoped for was a realistic estimate of the distribution of the probability of doses. The panel, with help from the contractor, could then set a safety factor by selecting a given probability, say 90%. As you are aware, I was uncomfortable with some of the behavioral parameters RAC selected. It was explained that the applicable guidance suggested using the 95% value for the behavioral parameters. While NUREG 1549 does recommend this approach for deterministic evaluations it specifically recommends actual distributions be used for probabilistic studies. At the time we discussed scenarios, I was assured that for the environmental parameters, the best scientific estimate would be used without additional safety factors. I was dismayed to see that for some of these parameters you made statements like "We feel that the use of this conservative value is reasonable, and will not be changed" or "while this is a conservative assumption, RAC will not change this value for our independent calculation because of the recognized importance of the inhalation pathway". In a quantitative risk assessment adding safety factors like these only serves to bias the result. To place safety factors on only the most important variables simply says if we are going to bias the result lets really bias it. If safety factors are to placed on the environmental parameters the resulting distribution of the doses will be biased, worse it will not be possible to quantify this bias. I would have difficulty in supporting any value other than the median from such a biased distribution of doses. What is really unfortunate is that for one of the variables that had the safety factor added, cover depth, the site data clearly shows that the correct value is zero therefore no safety factor is required. In the other cases there is ample scientific evidence for a site specific value therefore a safety factor is not required. The statements are therefore gratuitous but nevertheless do great harm to the study. They will tend to confuse the scientific reader and will provide powerful arguments with which to discredit the study. I ask that they be deleted.

We will delete any comments of this type. We appreciate this comment, and we address specific details below with regard to each individual parameter.

There are three specific parameters that I would like for you to review and comment on.

Indoor Dust Filtration

There is nearly a full page discussion on the External Gamma Shielding Factor, a parameter RAC admits has little effect on the RSAL, but only a short paragraph on the Internal Dust Shielding factor which RAC considers important. More disturbing is the RAC's justification for using the highest value: "While this is a conservative assumption, RAC will not change this value ... because of the recognized importance of the inhalation pathway". Are we to assume that the value chosen depends on it's importance to the calculation. How is this any better than a screening analysis. There would perhaps be some justification for the value used if a scientific value was not available. A casual examination of the literature revealed several studies that could be considered.

The RESRAD default is 0.4 following Alzona et. al. (1979). Harkonson and Kirchner (1996) in their critique of the RFCA RSAL values cited Romney and Wallace (1976) as supporting a value of 0.10. NUREG CR-5512 cites a IAEA publication as finding a substantial reduction in indoor dust levels vs outdoor levels. Schmel (1980) was also cited; he studied dust levels during various indoor activities including vigorous sweeping. A NRC draft report (1998) compares the approach in RESRAD to DandD. RESRAD simply scales the outdoor dust level while in DandD indoor dust levels are independent of the outdoor levels. This is following studies that show that most of the indoor dust levels are derived from indoor sources. The default indoor dust mass loading attributed to outdoor sources in DandD is 2.82x10-6 which in most cases is much less than the outdoor level. Lastly common sense would suppose that indoor dust levels are less than outdoor levels especially during the winter when the house is closed to outside ventilation.

After reviewing these studies I suggest that a value of 1.0 is not supported by the studies even at the screening level. I would suggest a normal distribution centered on 0.4 with a standard deviation of .15 truncated by 0.0 and 1.0.

We greatly appreciate these comments. They were quite helpful, and have caused us to take a second look at the indoor dust filtration.

Admittedly, the indoor dust filtration factor has the next greatest impact on the outcome of the calculation than any other parameter mentioned outside of the most sensitive parameters. It would also have a much greater impact on the calculation were the inhalation pathway in RESRAD V. 5.82 not minimized like it is, and this parameter will likely have an important effect on the final results of the *RAC* calculation.

Leaving the parameter at its DOE/EPA/CHPHE defined value was more a resources decision than anything else. We would like to spend a great deal of time defining what this parameter might be for different parts of the country, and specifically for Rocky Flats. There is a great deal of evidence that supports the use of a distribution to represent this value. We were at a place in the production of this report where the resources were better spent developing other parameters.

The comments on this parameter value, but particularly the comment from this reviewer, encourage us to look again at a possible distribution of values for this parameter. We feel that under unknown conditions, 1.0 is still a reasonable upper bound for this parameter. We don't yet have a feel for what a lower bound or median value might be, but

314

we have thought that an appropriate shape for the distribution might be skewed toward the higher end of the possible range (with the majority of the probability centered toward the high end).

We will continue to explore this parameter for the final version of the Task 3 report.

Area of Contaminated Zone

I had difficulty following your discussion of why the area of the contaminated zone is uncertain. You are correct that given the present contamination it is difficult to assign an area that is both homogeneous and includes the entire contaminated area. Your approach to the problem is novel and I believe it reveals many interesting insights into the origin and fate of the contamination coming from the 903 pad. As an estimate of the area of contamination I am less impressed. RESRAD assumes that the receptor is located at the downwind edge of the contamination. Given this assumption if the area of contamination includes large areas below the RSAL the dose to the receptor would be diluted and could result in estimating a lower than actual dose. If instead you think in terms of the maximum exposure to the receptor after the cleanup levels are met the problem is much easier. The cleanup should result in a large homogeneous area at a level below the RSAL. A problem with this approach is it is recursive, how do you find the area to be remediated before you determine the RSAL. As with many recursive problems this one converges. At least at Rocky Flats the area of contamination drops off rapidly with increasing radionuclide level. As a first assumption we could use the area for the RFCA Tier II residential Pu RSAL's which is 115 pCi/g. The area would then be about 120,000 m2 . I would use this value as the mean of a normal distribution with STD of 25,000.

We do not plan to use the RESRAD evaluation of receptor location and thus we will not use the RESRAD area of contamination. Because we are convinced that it is more meaningful to assess resuspension through use of the existing profile of contamination combined with the air concentration measurements, we need this profile. We hope this entire approach will become clearer through Task 5.

Distribution Coefficients

The groundwater pathway in RESRAD presents a dilemma to the modeler. If the pathway is to show any dose within the 1000 year modeling time the radionuclides must be mobile. If they are mobile then RESRAD shows they are rapidly leached from the soil resulting a decreased inhalation dose. In reality both may be contributors to the dose but the single parameter in RESRAD does not permit modeling this possibility. RAC has chosen a low value of Kd for Pu and Am based on the work of Dames and Moore (1984) in order to evaluate the groundwater pathway. The downside of this approach is it postulates a rapid decrease in inhalation dose. The distribution coefficient is normally thought of as a measure of the chemical leaching and movement of the soluble form of a radionuclide. More generally it can be thought of as a measure of mobility by any process including chemical, physical or biologic processes. The recent work of the Actinide Studies Group summarized in the present report on pages 7 and 8, indicates that chemical mobility probably is not important. RAC's excellent summary on pages 20 thru 25 of the present report presents a good basis for assuming that mobility in the top 30 cm of soil is controlled by a combination of biological and physical factors. Below 30 cm these processes seems to slow down. Litaor's new paper contains data to support that nearly all of the Pu transport at Rocky Flats occurs through flow of discrete particles or possibly colloids along localized shallow subsurface flow. This flow only occurs when the top several meters of soil have become saturated. He estimates that these conditions occurs about once every fifteen years. Under these conditions the movement is lateral and follows topography. The RESRAD groundwater model is completely useless to handle these conditions. Based of the best data available the Actinide Studies Group has made a preliminary estimate that the Kd is between 10,000 and 100,000 with 20,000 being the most likely value. I would recommend that RAC examine the groundwater pathway separately from the base case. For the base case a lognormal distribution with a geometric mean of 20,000 could be used for Pu. I don't have a suggestion for Am but it is probably over 10,000 cm3/g.

Lateral movement of actinides may be important in fact it may determine the cleanup levels. I am not suggesting eliminating the pathway on the contrary it is too important to use a false and simplistic model like RESRAD. Your preliminary work shows groundwater contamination becoming a problem in several hundred years; I believe it is a problem today. I suggest using a qualitative model like Litaor's to give some early warning of what to expect.

Based on the comments of this and other reviewers, we will evaluate the residential rancher both with and without the groundwater pathway, to provide some indication of the impact that this pathway might have on dose. It is true, as this reviewer points out, that the groundwater pathway within RESRAD presents a dilemma. It is clear that within the context of this study, the details of this pathway cannot be worked out, but can be at least qualitatively evaluated for direction of future studies.

We are examining the referred to document to better assess a distribution for K_d.

References:

Alzona J. et. al. 1979, "Indoor-Outdoor Relationships for Airborne Particulate Matter of Outdoor Origin", Atmospheric Environment 13:55-60.

Dames and Moore 1984. De Minimus Water Impacts Analysis Methodology. NRC study

Harkonson T. E. and Kirchner T. B. (1996) Oral report to the RFCAB Spt 9, 1996

IAEA, 1970, Monitoring Radioactive Contamination on Surfaces. Technical report Number 120

Litaor I., Barth G., Litus G. 1999, The Hydro-geochemistry of Actinides in the Soil of Rocky Flats, Colorado, manuscript.

NRC Draft Report 1998, Comparison of the Models and Assumptions used in the DandD 1.0, RESRAD 5.61 and RESRAD-Build Computer Codes with Respect to the Resident Farmer and the Industrial Occupant Scenarios provided in NUREG/CR-5512.

Romney and Wallace (1976) I don't have the reference.

Schmel G. A. 1980, Particle Resuspension: A Review, Environ. Int. 4:107-127

3\10

Joe Goldfield

Portions of the task three report are very troubling. One cited soil action level, resulting from the application of RESRAD to Rocky Flats open space, is 53,120 pCi/g (picocuries of plutonium per gram of soil) well over1,000,000 times as high as the average plutonium background level (0.04 pCi/g).See page three comparing the RESRAD version 5.61 to RESRAD version 5.82. The last column shows an action level of 53,120 pCi/g of soil for the open space and 8351 pCi for a resident with a dose level of 85 mrem/yr.

We intended this presentation to serve only as an illustration of why we have chosen to bypass the resuspension calculation in RESRAD. These values have spurred so much comment that we plan to consider reworking this entire section of the report to include only a discussion of the different versions and not to present tables of values extracted from the versions.

1. The definition of TRU (transuranic waste) that must be sent to WIPP includes materials that contain greater than 100 nCi of plutonium per gram of waste. The cleanup standard for the open space would be over 53 nCi of plutonium per gram of soil (halfway up to the TRU waste designation). Furthermore in accordance with the report on Sampling Protocols, hot spots that may be ten times the cleanup standard (530 nCi Pu/g) would not be cleaned up. Thus areas could contain over five times the lower limit of TRU waste. In accordance with this thinking why would we send anything to WIPP and bury it 2,000 feet underground? If we played our cards right we could spread it around the open space.

Bear in mind that 530 nCi/g is equal to 8413 ng (nanograms of plutonium) (the concentration given in nCi/g must be multiplied by 15.9 to convert to ng of Pu/g) or 8.4 ug (micrograms of plutonium) per gram of soil while the allowable lifetime body burden of a nuclear plant worker is only 1ug. The ingestion or inhalation of a little over a tenth of the soil concentration would exceed the allowable lifetime body burden of a nuclear plant worker.

We did not present the value cited here as a possible soil action level. We presented it only to show how inadequate we believe the resuspension code in RESRAD to be for predicting possible soil action levels and why we believe it to be necessary to prepare our own calculation. We apologize for any confusion this might have caused.

2. On page 39 RAC cites the ingestion of soil at the 95 percentile level as 0.75 g per day. With a level of 8.4 ug of plutonium per gram of soil in the hot spots of the open area--the rate of ingestion would be 6.3 ug/day or 6.3 times the allowable lifetime body burden of a nuclear plant worker. If we place a safety factor of ten or twenty for civilians and children, every day of soil ingestion of hot spots results in ingestion of 60 to 100 times the allowable lifetime plutonum body burden.

Again, the values cited for mass of plutonium in soil are based on results that were presented for illustrative purposes only.

3. Examine the soil action level allowable for residents of the remediated portions of Rocky Flats where the soil action level is 8351 pCi/g which is equal to 8.35 nCi/g (nanocuries per gram

of soil). Converting to nanograms requires multiplication by the factor of 15.9 giving 133 ng of Pu per gram of soil.

RAC states that the data extropolating from soil concentrations to inhalation quantities is meager. My information is also meager. Permit me to use my methods of estimating.

I have seen data that shows that the plutonium in soil is concentrated in the small particle size fraction. Air blowing over the soil would tend to most easily entrain the smaller particle size fraction of the soil. It is reasonable to guess that air borne soil has 3 to 5 times the soil concentration or 400 to 670 ng of plutonium per gram of soil.

If a person breathes 10,000 cu. meters of air per year and the particulate concentration is 90 ug per cubic meter (instead of the 26 discussed in the report), the yearly particulate intake will be 900,000 ug or 0.9 g of soil. That soil would contain 360 to 600 ng of plutonium 9 (30 to 50 ng per month). I have mentioned previously that the allowable lifetime body burden of a nuclear plant worker is 1 ug or 1000 ng (nanograms). Assume a reduction of tenfold for the general population--that allowable body burden would fall to 100 ng. It would take two to three months of residency to exceed the allowable body burden.

This result assumes the average concentration of 8351 pCu/g rather than the probable effect of pockets of contamination that far exceed the average.

Again, the values cited for mass of plutonium in soil are based on results that were presented for illustrative purposes only. We continue to consider revisions to this section of the report to eliminate the ability to make any inappropriate comparisons or calculations with these results, which are not results of this study.

4. The area of the contaminated zone is estimated as 40,000m2. That is 200 meters by 200 meters. That is only 660 feet square. That area is tiny compared to the total plant area which amounts to thousands of acres. The area probably does not include the industrial area which may have ten times the plutonium contamination of the 903 pad. For some reason the discussions of plutonium contamination are restricted to the 903 pad and do not include the industrial area.

This area is not suggested as the area to be used for the new soil action level calculations. This area was used in the previous analysis.

5. I suggest that we have a knowledgeable expert on Rocky Flats meteorology review the meteorological data presented. Gale Biggs in previous reviews took exception to much of the data available at Rocky Flats.

The data presented in this report are from recent (1989-1993) meteorology reports at Rocky Flats. These data have been used in other projects completed at Rocky Flats and we are confident in their ability to predict annual average wind conditions at the site.

6. I have taken exception in the past to the use of 26ug per cubic meter as the particulate concentration in air at Rocky Flats. I understand that that particulate concentration is based on measurements taken by means of high volume PM10 samplers located at Rocky Flats. My reservations are based on the following:

318

a. PM10 samplers remove 50% of the airborne particulate concentration. Some significant percentage of the material removed is smaller than 10 microns and is therefore in the respirable range.

b. PM10 samplers must be carefully handled to get acceptable data. They must be calibrated so that the exhausted air volume is known accurately. Account must be taken of the pressure buildup on filters and the resultant reduction in flow.

c. The location of the samplers, I surmise, are on the periphery of the property where the site resembles wilderness areas instead of heavily populated and developed areas that may result in the future at Rocky Flats. Our analyses must allow for the foreseeable changes that will occur at Rocky Flats over the next 1,000 years

d. Does the RESRAD program correct the particulate concentration entered into the calculations to reduce the total particulate to account for fractions that may be larger than the respirable sizes? If so, using PM10 results may introduce a double particulate reduction to account for non-respirable size particles.

e. For all the reasons stated and the fact that a consultant reporting to the RFCAB recommended an airborne soil particulate concentration of 90ug, I strongly recommend that the estimated particulate concentration be raised to 90ug per cubic meter.

We plan to derive a resuspension factor/mass loading value from available site-specific data and undertake a calculation of resuspension from this factor independent of the RESRAD calculation.

7. I have not had the time to investigate the subject of breathing rates which I still believe are not being estimated conservatively.

Joel Selbin

I want to see a really detailed explanation of why RESRAD 5.82 yields considerably higher SALs (page 3 and Appendix A) than RESRAD 5.61. The statement on page 2 that the former version of the code used a "conservative treatment" of the very important matter of resuspension is very disconcerting. What other factors are going to have a comparable effect, and in which direction? What happens to SALs at other world sites using the new code?

The comments resulting from the inclusion of this table comparing the results of the two versions of RESRAD are numerous. As stated in response to the previous reviewers' comments, we are considering completely rewriting this section to better reflect the intent of including it in this report.

The documentation that accompanies the newer versions of RESRAD state that the previous treatment of resuspension was conservative and generic. Because the current treatment is still unsatisfactory to *RAC* and appears to produce significantly higher soil action levels and lower doses, we plan to not use the newer version's treatment of uncertainty.

We do not have the resources of the time to review the impact of this code at other sites, and in fact, it is unnecessary given the intent of this presentation: to impress on the panel the importance of the treatment of resuspension that *RAC* is undertaking.

320

LeRoy Moore

Where do "Relative Biological Effect" numbers for Pu appear in the RSAL calculations? Are they among the inputs and assumptions about which the assumption is made that they do not modify the outcome? If so, I will make a comment on them. If not, when will they be considered? This is an issue about which I pressed hard but to no effect with the government agencies when they adopted the original RSALs.

Relative biological effect is built into risk assessment, which in turn is built into the dose limits provided for this study (15 and 85 mrem y^{-1}). We plan to comment on risk in the Task 5 report in terms of what it means in the context of this study.

p. v, Ex. Sum: about three-fourths down in the opening paragraph a sentence begins: "As a result of public concern about the proposed soil actions levels. . . ." Delete "proposed" and change to read: "soil action levels adopted in Octorber 1996."

We will consider this change to the text.

p. 1: Change opening sentence of Intro to read: "Soil action levels are calculated to identify the concentration of one or more radionuclides in the soil above which remedial action would be required to prevent people from receiving doses above an officially designated level."

We will also consider this change.

pp. 1-3: Why is RAC using RESRAD 5.82 rather than 5.61? My recollection is that at one meeting a couple of months ago RAC presented us with the disturbing info that 5.82's parameters had been so modified that feeding in the same data used by the agencies in setting the original RF RSALs resulted in much higher allowed concentrations of Pu, etc. The text on pp. 2-3 (esp. Table 1 on p. 3) repeats this info. We go from a RSAL for Pu of 1429 pCi/g to one of 8351, which, to put it mildly, is outrageous. I do not recall that the Panel asked RAC to proceed with 5.82. I do recall that there was a request for documentation from DOE of the instructions they gave to Argonne along with their request that RESRAD be updated. Have we received this documentation? Short of getting it and thus understanding why the outcome from calculations is so much higher on the revised RESRAD, I think we should stay with the program used by the agencies initially. Is there any reason we cannot do this?

We used the newer version of RESRAD because, at the outset of this project when we requested source code and documentation, we received source code for Version 5.82. At the end of it all, however, it matters not what version of RESRAD we use as long as it is understood that the resuspension calculation, the only major change in the updated version of the code, will be bypassed for this assessment in favor of a site-specific resuspension model. It is too late in the project to make any changes in the code selected for use, and it is not necessary, given what we plan to do about resuspension.

p. 2, second para. under "Difference between versions": Why use a value for annual mean for Denver area wind speed derived from a National Climatic Data Center report? Isn't there site-specific data for wind speed at RF? RAC may recall that wind is stronger at RF than in Denver, and that the prevailing wind blows in a different direction. The RF original siting resulted from a mistake about wind, namely, that it was based on readings done in Denver, not at RF itself.

We present this data because it gave us a place to begin our sensitivity analysis for Version 5.82. We plan to use data originating from the Rocky Flats Meteorological Station from the years 1989-1993 to make our calculations. These data were available in the appropriate format and therefore ready to use.

p. 8: Contrary to what is said in the first full paragraph, Litaor thought he found Pu in particle and colloidal form moving with groundwater in May/June 1995. He at least speculates, as I understand his work, that anoxic conditions of soil saturation may release some Pu into dissolved form. The second full para. on this page refers to this aspect of Litaor's work, but I wonder if it's correct to suggest that subsurface storm flow could be important only for "localized soil contamination areas," since seeps release material into stream channels that go to holding ponds or eventually exit the site. Also, it's not clear that channels have been adequately analyzed in terms of their ability to hold material flowing throught them; that is, do they leak?

We will review this section of the report to ensure that it is consistent with the literature.

p. 30: My note above about wind may be answered from RAC's perspective on pp. 29-30. But I raise a further question regarding RAC's assumption that "high winds will not be explored further in the SAL project." Why? Evidently because wind blows contamination away and thus lessens possibility of future resuspension by this means. OK. This makes sense, though it's not very reassuring news. But a decision to set aside further analysis re. wind seems predicated on the assumption that the 903 Pad will not release more and that main sources of resuspension have been already depleted. What about remediation of 903 area? What about taking down of buildings and exposure of whole new areas of contaminated soil? What about any construction activities that may occur? There seems to be ample reason to keep airborne resuspension alive as a very likely pathway for future exposure of unwitting populations. Am I missing something here?

We do not intend to eliminate the airborne resuspension pathway. The intent of this section of the report is to respond to the often heard comment about the severity of the high winds at Rocky Flats. It is true that wind speeds at Rocky Flats and in general along the Front Range in Colorado can reach very high speeds. What was learned in the dose reconstruction project, however, is that although high winds tend to resuspend a great deal of material, that material is generally dispersed rapidly. This rapid dispersion decreases the air concentrations at close to the source locations and thus decreases the dose to individuals that are of interest for this project. For that reason, we will not consider high winds, but rather average Rocky Flats winds resulting in resuspension.

p. 31: Re. scenarios, one peer reviewer in commenting on Task 2 raised a serious question re. "institutional controls." In a May 7, 1999, memo to RAC I raised the issue as follows: "One of the peer reviewers for the independent assessment of the Rocky Flats RSALs states that the

322

RSALs as adopted misapply the concept of 'institutional controls' in relation to the 15/85 mrem/year dose (see attached 'Review Comments on the March 1999 Draft Report . . . for Task 2: Computer Models," section 1, 'Application of the 85 mrem/y criterion'). This suggests that the Rocky Flats RSALs violate CERCLA in the way the 'institutional controls' concept is employed. What corrections need to be made?" I raise this question anew because it was not previously answered and because it comes up again under "scenarios." One of the scenarios included in the officially adopted RSALs – the hypothetical future resident – assumes disappearance of institutional controls, in possible violation of CERCLA, if the peer reviewer is correct in the comment submitted. If the reviewer is correct, then the hypothetical future resident scenario (as well as all other hypothetical future scenarios) needs to be recast in terms not of a possible dose of 85 mrem/yr but of 15. How does RAC respond?

This same reviewer brought up this topic again. We respond by reminding the panel that we will present distributions of soil action level for both dose criteria for all scenarios. The panel and *RAC* can then work together to develop recommendations to DOE.

pp. 34-36: This section does not make sense to me. Table 11 shows breathing rates ranging from 7.5 L/min to 712. Is this correct? The numbers given on p. 36 seem far less than those provided by Joe Goldfield January 31, 1999, paper. Joe's paper has the virtue of clarity and persuasiveness. I defer to him in the hopes he will make a clear response to this section.

Thanks to the reviewer for noting this typographical error; a hyphen was missing and it should read 7–12. The appropriate change will be made in the final report. The breathing rate distributions shown in Figure 6 in the report were those the panel agreed upon at the May 1999 meeting, following several months of intense panel discussion and the consultation with a specialist in respiratory physiology at CSU.

pp. 37-40: Re. soil ingestion, I again defer to Joe Goldfield.

As with the breathing rate distributions, the distribution of soil ingestion rates and the selection of the value for use in the scenarios was approved by the panel at the May 1999 meeting. We considered many published reports, along with Joe Goldfield's paper he wrote for the panel, in our assessment.

DOE COMMENTS

Comments and Questions on *RAC*'s Draft Report for *Task 3: Inputs and Assumptions*

1. Pages 4 through 10 of the draft Task 3 report summarizes the results of a sensitivity analysis, but does not provide the full documentation that lies behind this analysis. At the RAC Sensitivity Analysis for RESRAD Parameter presentation on January 14, 1999, the most sensitive parameters were identified as solubility of plutonium/dose conversion factor and the mass loading factor. The less sensitive parameters were identified as cover depth, breathing rate and soil ingestion. During the Project Update presentation in May 1999, the impacts between using RESRAD v5.61 and 5.82 were identified. The documentation supporting the sensitivity analysis is needed to understand how RAC classified the parameters as discussed on page 4 of the Task 3 Report without having an independent reviewer repeating each sensitivity analysis. Please provide in the final report documentation supporting the sensitivity analysis.

We will include a more detailed discussion of the sensitivity analysis in the final version of the Task 3 report.

2. RAC has recommended an "Indoor Dust Filtration" factor of 1.0 (page 5). The Rocky Flats Cleanup Agreement (RFCA) Parties have identified new information from both EPA (Exposure Factors Handbook) and NCRP (NCRP Report No. 129) that may impact this input and are evaluating this information as part of the RFCA annual review process. Has RAC evaluated the new information available from the EPA and NCRP as it relates to this parameter?

We are exploring a distribution of values for the final version of this report as a result of the significant number of comments on this parameter. We thank this reviewer for identifying additional documentation to assist us in this task.

3. Table 2, "Relative Concentration of Radionuclides in Soil at Rocky Flats in 1999," could not be verified with the information and references provided in the draft report. Please include in the final report the data representing how the mass values from the references listed were converted to activities and allowed to decay (or grow in, in the case of ^{241}Am) to the year 1999 for use in the RESRAD calculations.

Because this reviewer could not reproduce the values in Table 2, we will review the calculations to ensure that they were done correctly. The conversions from mass to activity were done using the latest available specific activity values, decay occurred via radioactive decay (using the latest available half-life values) and including a generic weathering constant of 4.0×10^{-4}.

4. It is not clear from the Task 3 report how RAC plans to analyze the agency scenarios. Specifically, it is not clear if RAC plans to substitute its own parameter values for the agency values (as shown in Table 4) in calculating new recommended RSALs for the agency scenarios. Can RAC clarify this issue? Also, Table 10 lists the different Scenario Parameter

324

Values for DOE and RAC scenarios. It is not clear from the table or from the text if RAC concurs with or is simply not analyzing the parameter values for the DOE scenarios. For example, the agencies assumed for an Open Space scenario a value for time on site of 125 hours per year. By not adjusting this parameter, is RAC endorsing it or simply choosing not to analyze it? Or has RAC concluded that it is not sensitive and therefore does not merit more detailed analysis? In other words, does RAC agree that the agencies have appropriately defined their own scenarios, or for the purpose of analysis is RAC simply accepting the Scenario parameter values as is?

We plan to analyze the agency scenarios and the *RAC* scenarios using the scenario parameters presented in Table 10 and the site-related parameters presented in Table 4 (*RAC* value column for all scenarios) and the accompanying text. The agency scenarios were in close agreement with similar *RAC* scenarios that were previously developed but subsequently dropped because of their close resemblance to the agency scenarios. The determination of the scenarios by which to evaluate soil cleanup levels is to be made by the panel after presentation of results of the analysis for all scenarios.

5. The Actinide Migration Team has recently completed work directly related to Kd values. We attached a copy of the report that we believe is relevant to the Task 3 report.

Upon receipt of these comments, we requested and have received a copy of this report. We thank this reviewer for bringing this report to our attention and plan to evaluate it and possibly incorporate the results for the final version of this report.

6. RAC has defined a model of ^{239}Pu concentration in soil as a function of location (page 20). Do similar models need to be defined for ^{241}Am or U? If yes, what task report will explain this extrapolation? If not, will the Pu data be extrapolated for Am and/or U?

Americium and uranium concentrations will be extrapolated from this model based on the radionuclide ratios given in Table 2.

7. Figure 2 represents the locations of more than 588 soil samples of ^{239}Pu at Rocky Flats which were used as a basis for a spatial model. While the text states the sources of the raw soil concentration data, the text also states that the 588 soil samples are a subset of the raw soil concentration data (page 22). Please provide in the final report a list, including the source, of the 588 entries.

The database of the soil samples used to create this distribution was defined for the Phase I dose reconstruction project, and is outlined in the ChemRisk Task 6 report (1994). Additionally data was needed to supplement this historical database, and those data were obtained from the data set deposited by M.I. Litaor with the Colorado Department of Public Health and Environment. These supplemental data were used to enhance the resolution of measurements available at locations near the 903 Area.

8. RAC's recommended breathing rates (page 36) could not be verified with the information in this report. As captured in the RAC Scenario presentation on January 14, 1999, it is

important to understand the duration of daily activities for each receptor in order to calculate a breathing rate. For clarity, please incorporate the assigned duration for the various daily activity levels in the final report. Also, please incorporate the distributions of breathing rates for active and sedentary adults, for active and sedentary children, and for active and sedentary infants (as captured in the RAC Breathing Rate Distributions presentation on March 11, 1999) in the final report. Please also explain why and on what basis RAC recommended using the 95th percentile value from the breathing rate distribution.

The selection of breathing rate values for the scenarios was a long process involving many discussions with the panel and consultation with a respiratory physiologist. In developing our breathing rate distribution we reviewed numerous reports as described in the Task 3 report. We did develop detailed breakdown of time/activity levels for each scenario and have that information available. We will consider the reviewer's request to include those detailed spreadsheets in the report.

9. RAC recommended identical annual soil ingestion values for each of RAC's recommended scenarios, i.e., current site industrial worker, resident rancher, infant of rancher, and child of rancher (page 39). Is it possible to create a frequency distribution of soil ingestion values for each scenario similar to what was done for breathing rates?

We did create a distribution of soil ingestion across the population, but based on the types of information available on soil ingestion, it was not reasonable to create the same type of frequency distribution based on scenarios.

10. The RAC recommended consumption rates for fruits, nonleafy vegetables and grains (page 40) could not be verified from NCRP Report 129. Please state where in NCRP Report 129 these ingestion rates were taken. There is currently no reference for the RAC recommended leafy vegetable consumption rate.

We will make the appropriate revisions in the report so that the source of these values is clearly referenced.

11. RAC states on page 27 of the draft Task 3 report that monitoring data do not provide particle size information. Since 1995, the Kaiser-Hill Team has been reporting, in the Quarterly Environmental Monitoring Report, air monitoring data from selected locations and time periods at the Site that contain size-segregated radionuclide concentrations, separated at about 9 to 10 micrometers. Has RAC evaluated this information as it relates to this parameter?

This information was not available to use at the time the production of this report was completed. We would like to receive this information, but it is not clear that we would be able to use it in the final modeling effort for this project, which is already well underway.

326

FINAL REPORT

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

October 1999

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

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327

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

FINAL REPORT

Task 3: Inputs and Assumptions

Radionuclide Soil Action Level Oversight Panel

October 1999

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

328

EXECUTIVE SUMMARY

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

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

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

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

Most Sensitive Parameters

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

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

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

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

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

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

The term mass loading was used in this analysis as a measure of resuspension of soil from the ground. Resuspension is a complex process that is affected by many environmental factors

330

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

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

Less Sensitive Parameters

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

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

For these somewhat sensitive parameters *RAC* used the values from the DOE/EPA/CDPHE assessment for cover depth and indoor dust filtration. For the other four, values more consistent with studies published in the open scientific literature were selected. For the depth of soil mixing layer, or the depth over which soil is uniformly distributed, *RAC* selected a value of 0.03 m, instead of 0.15 m, based on published studies at Rocky Flats. For the thickness of the contaminated zone, *RAC* selected a value of 0.20 m, instead of 0.15 m, based on studies that show the contamination is distributed over the top 20 cm (0.20 m) of soil with very little movement of the contamination over the past 20 years. For the fraction of irrigation water contaminated by groundwater (irrigation water contamination fraction), *RAC* determined that groundwater might be used for irrigation or as a source of drinking water. As a result, they assumed that all of the groundwater used for irrigation would be contaminated (irrigation contamination fraction = 1.0).

In the previous assessment, it was assumed that none of the water would be contaminated (irrigation contamination fraction = 0).

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

Other Parameters

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

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

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

Scenarios

The Task 3 report describes the seven scenarios that are currently being evaluated: the three scenarios described in the previous assessment, *Action Levels for Radionuclides in Soils for the*

337

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

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

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

CONTENTS

FIGURES

335

TABLES

TASK 3: INPUTS AND ASSUMPTIONS

INTRODUCTION

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

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

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

Parameters Explored

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

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

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

Difference between Versions of RESRAD

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

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

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

338

SENSTIVITY ANALYSIS

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

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

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

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

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

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

Sensitive Parameters

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

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

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

Parameters with Limited Sensitivity

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

Cover Depth

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

Depth of Soil Mixing Layer

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

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

In the previous soil action level calculations (DOE/EPA/CDPHE 1996), the values for soil mixing layer and thickness of the contaminated zone were equal. *RAC* did not believe that setting the available depth for resuspension and the total thickness of the contaminated zone equal to each other was supported by the data from Rocky Flats. Based on the research of Webb et al. (1997), *RAC* selected a value of 0.03 m for the depth of the soil mixing layer. We were, in fact, constrained to the use of this depth by the available soil concentration data.

340

Soil-to-Plant Transfer Factors

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

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

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

[a] Source: NCRP (1999).

Indoor Dust Filtration

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

Irrigation Water Contamination Fraction

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

Thickness of Contaminated Zone

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

Parameters not Exhibiting Sensitivity

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

External Gamma Shielding Factor

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

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

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

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

342

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

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

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

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

Initial Concentration of Radionuclides

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

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

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

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

The current value for ^{239}Pu contamination varies spatially. *RAC* has identified contours of contamination levels using soil concentration data from Litaor et al. (1995), Litaor and Zika (1996), Webb et al. (1997), Illseley and Hume (1979), Ripple et al. (1994), Krey et al. (1976), and the CDPHE. We develop and present these contours in a later section of this report.

Uranium concentrations are more difficult to determine. Available data suggested that uranium exists on the Rocky Flats site in a few small "hot spots." Determining where those hot spots might exist within the scope of this study is difficult.

Litaor (1995) looked at the extent and distribution of uranium in the Rocky Flats environment. Litaor discovered that the uranium followed no recognizable spatial distribution pattern and was not in concentrations readily discernible from background, for the most part. The RFP contribution to ^{234}U was determined to be negligible. The elevated soil concentrations of ^{235}U were localized to an area east of the industrial section. Litaor suggested that these concentrations might have resulted from surface flow and interflow from the east spray field (Litaor 1995). The ^{238}U activities that were the highest were located in the immediate vicinity of the 903 Area, but they did not extend beyond that area, suggesting that uranium was not dispersed in the same way as plutonium. Litaor suggested that this is likely due to the differences in the solubility characteristics of the two nuclides.

Even with the few elevated concentrations of uranium, the concentrations of ^{234}U and ^{235}U were generally well within the natural range for uranium isotopes in soil. Only ^{238}U showed elevated concentrations in Litaor's study area, and those were located immediately around the 903 Area.

It is likely that the most significant uranium concentrations would exist in locations where uranium was stored or burned, such as the trenches, or perhaps in solar pond sediments. Uranium contamination is definitely site-specific and would be above background only at a limited number of locations as dictated by Rocky Flats operations and disposal practices. Certainly, uranium is not distributed in any recognizable spatial pattern, and uranium contamination probably only exists in hot spots. The extent, concentration, and location of these hot spots are important for calculating any contribution to dose from uranium.

For our calculations of soil action levels for uranium, we selected a single location for which concentrations might be at a maximum and determined an action level for that location. This guides us to a better understanding of uranium and its potential risk to those at the Rocky Flats location. These calculations will be accomplished and outlined in the Task 5 report.

344

Plutonium Solubility and Dose Conversion Factors

Results from ongoing Actinide Migration Studies (AMS) at the site are helping to characterize the chemical and physical form of plutonium at the Rocky Flats site. The plutonium that is found in Rocky Flats soil is generally highly insoluble and attached to soil particles. This view is supported by the AMS, which show the effectiveness of the retention ponds in removing suspended solids and associated plutonium (and americium) from site surface water (RMRS 1998). Much of the plutonium discharged to Pond C-2 settles out of the water column, and plutonium concentrations measured further downstream in Woman Creek are an order of magnitude lower. In contrast, the ponds are less effective at removing uranium from the water column. This is expected because uranium has a higher solubility than plutonium and is more susceptible to dissolution and transport in the solution phase.

Recent work by researchers at the Los Alamos National Laboratory has characterized plutonium in samples from the 903 Area. Using powerful, new state-of-the-art analytical techniques, they have demonstrated that plutonium from under the asphalt pad at the 903 Area is insoluble PuO_2. The plutonium/americium ratio also indicates insoluble plutonium. These new results tend to confirm that plutonium in the soil at Rocky Flats is insoluble PuO_2 and, thus, may not get into the groundwater. While results from some of the AMS indicate that this insoluble form of plutonium may not enter groundwater, we are including the groundwater pathway in the rancher scenario. We do recognize, however, that our assessment of the groundwater pathway is limited by the pathway's complexity.

Plutonium mobility is another area under investigation by the AMS researchers that may play an important role at the site. One situation that may result in increased plutonium mobility is during extraordinary precipitation events in which the soil is saturated for significant amounts of time (Litaor and Zika 1996). Such conditions may result in subsurface storm flow, which is rapid, saturated, near-surface lateral flow from hill slopes that can discharge to seeps and streams because the groundwater is moving rapidly at a shallow depth. Subsurface storm flow is a potentially important pathway for plutonium in localized surface soil contamination areas where shallow or perched groundwater discharges to seeps or stream channels.

These solubility studies allow dose conversion factors to be determined for plutonium and other radionuclides. Insoluble forms of plutonium would be classified as slow clearance materials. In ICRP 30 (ICRP 1978), these forms of plutonium were classified as clearance type Y. *RAC* has researched the most updated values available for dose conversion factors from ICRP (1999). Clearance classification has changed somewhat. Instead of identifying clearance based on time it takes to clear the material (D, W, or Y to represent days, weeks, or years), ICRP identified the clearance by rate at which material is cleared (F, M, or S to represent fast, medium, or slow). These classifications are generally interchangeable on a respective basis, so insoluble plutonium would now be classified as type S. Table 3 shows the most recent values for inhalation and ingestion dose conversion factors in comparison to the values from ICRP 30 for the radionuclides of interest at Rocky Flats.

Table 3. Dose Conversion Factors (DCFs) for Independent Calculation (mrem pCi^{-1})[a]

Radio-nuclide	ICRP 30[b] clearance class	ICRP 30 Inhalation DCF	ICRP 71[c] clearance class	ICRP 71 Inhalation DCF	ICRP 30 f_1	ICRP 30 Ingestion DCF	ICRP 67[d] f_1	ICRP 67 Ingestion DCF
^{241}Am	W	0.444	M	0.155	0.001	0.00364	0.0005	0.00074
^{238}Pu	Y	0.288	S	0.059	0.00001	0.0000496	0.0005	0.00085
^{239}Pu	Y	0.308	S	0.059	0.00001	0.0000518	0.0005	0.00093
^{240}Pu	Y	0.308	S	0.059	0.00001	0.0000518	0.0005	0.00093
^{241}Pu	Y	0.00496	S	0.00063	0.00001	0.00000077	0.0005	0.00002
^{234}U	Y	0.132	S	0.035	0.05	0.000283	0.02	0.00018
^{235}U	Y	0.123	S	0.031	0.05	0.000267	0.02	0.00017
^{238}U	Y	0.118	S	0.030	0.05	0.000269	0.02	0.00017

[a] The units of mrem pCi^{-1} are the conventional units used in RESRAD. To convert to standard units of Sv Bq^{-1}, simply divide the value in the table by 3700.
[b] ICRP 30 values have been used in RESRAD Versions 5.61 and 5.82.
[c] ICRP 71 listed the latest inhalation dose conversion factors (also given on ICRP CD-ROM [ICRP 1999]).
[d] ICRP 67 listed the latest ingestion dose conversion factors (also given on ICRP CD-ROM [ICRP 1999]).

Dose conversion factors do exhibit some limited age dependency. For very young babies (0–3 months), f_1[a] values for ingestion are as much as 10 times higher than the adult values, increasing the dose conversion factor by about 16 times. All other ages have ingestion dose coefficients somewhat less than a factor of 2 higher than the adult values.

The dose conversion factor values have changed rather significantly since the last ICRP publication. There are a number of reasons for these changes.

For inhalation dose conversion factors, changes in the respiratory tract model have the largest effect on the differences. The new respiratory tract model indicates reduced uptake from the lung. For an aerosol with an activity median aerodynamic diameter of 1 μm, the new model indicates roughly 50% of the inhaled activity deposited in the tract, in contrast with the 63% predicted by the old model. This distinction results from the new model being characterized as a nose breather, where a large fraction of the inhaled activity would be deposited in the anterior regions of the nasal passage and would never make it to the gastrointestinal tract to be adsorbed. The difference in deposition between the two models is almost a factor of two.

There is also a new model for the fraction of the lymph node irradiation attributed to lung dose, as well as a new model for the behavior of plutonium once it enters the blood stream, considering the movement of plutonium from bone surfaces into bone volume. All of these factors contribute to lowering the absorbed dose from inhalation of unit activity of plutonium.

The ingestion dose conversion factors reflect the difference introduced by the changes in the behavior of plutonium in the blood stream, as well as differences in new tissue weighting factors and adsorption coefficients (Eckerman 1999).

[a] f_1 is a factor that defines the retention of radionuclides in the body. The higher the value of f_1, the greater the retention.

346

Remaining Parameters

The outcome of the calculation was not sensitive to changes in the following parameter values:

- Nearly all of the saturated zone parameters (excluding K_d)
- All of the uncontaminated zone parameters
- Nearly all of the contaminated zone parameters including evapotranspiration coefficient, erosion rate, porosity, conductivity, density, b parameter, precipitation, irrigation rate and mode, and runoff coefficient
- Length parallel to the aquifer
- Watershed area
- Storage times for food
- Mass loading for foliar deposition
- Plant contamination fraction
- Thickness of the unsaturated uncontaminated zone
- Water table drop rate
- Well pump intake depth
- Well pumping rate.

Because of the insensitivity of the calculation to changes in these parameter values, we determined that additional work characterizing these values was not justified. In all cases, we accept and will use the values suggested in the original soil action level document (DOE/EPA/CDPHE 1996). In two cases, DOE used different values for the same parameter in each of the three scenarios in the existing soil action level calculations (DOE/EPA/CDPHE 1996). These parameters were irrigation rate and evapotranspiration coefficient. Neither of these parameters were found to be very sensitive to change. *RAC* used the values selected in the DOE/EPA/CDPHE calculations for the hypothetical resident scenario (DOE/EPA/CDPHE 1996).

Some of these remaining parameters that were not sensitive to change are part of the drinking/groundwater calculation, and they have no impact on the current soil action level calculation (DOE/EPA/CDPHE 1996) because none of the scenarios include the drinking water pathway. We explore the impact of this pathway in the following section of this report. Table 4 compares the parameter values to be used in the independent calculation to the DOE/EPA/CDPHE values. For more information on the distributions and values for the sensitive parameters, refer to the section titled "Uncertainty Distributions."

Table 4. Parameter Values to be Used in the Independent Calculation

Parameter name	DOE value	*RAC* value
	Sensitive Parameters	
Distribution coefficient	Pu = 218 cm^3 g^{-1} (or L kg^{-1}) Am = 76 cm^3 g^{-1} U = 50 cm^3 g^{-1}	Treated stochastically based on Rocky Flats measurements and other available data
Area of contaminated zone	40,000 m^2	Defined based on soil concentration measurements
Mass loading	0.000026 g m^{-3}	Model calibrated based on results of soil and airborne concentration analysis
Mean annual wind speed	Not required for RESRAD V 5.61	Used annual average wind data collected over 5 years
	Limited Sensitivity Parameters	
Thickness of contaminated zone	0.15 m	0.20 m
Inhalation shielding factor	1.0	1.0
Soil-to-plant transfer factors	Deterministic Pu = 1.0×10^{-3} Am = 1.0×10^{-3} U = 2.0×10^{-3}	Treated stochastically based on NCRP 129 recommendations
Cover depth	0 m	0 m
Irrigation water, contamination fraction	0	1.0
Depth of soil mixing layer	0.15 m	0.03 m
	Parameters Not Exhibiting Sensitivity	
Initial concentrations of radionuclides	100 pCi g^{-1}	Based on soil concentration measurements by Webb et al. (1997), Litaor (1995), Illsley and Hume (1979), CDPHE (as deposited by Litaor), and Krey et al. (1976)
External gamma shielding factor	0.8 – residential 0.014 – open space 0.17 – office worker	0.7 – for all scenarios, indoor/outdoors time fractions will describe occupancy
Density of contaminated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Contaminated zone erosion rate	0.0000749 m y^{-1}	0.0000749 m y^{-1}
Contaminated zone total porosity	0.3	0.3
Contaminated zone effective porosity	0.1	0.1
Contaminated zone hydraulic conductivity	44.5 m y^{-1}	44.5 m y^{-1}
Contaminated zone b parameter	10.4	10.4
Evapotranspiration coefficient	0.253 – residential 0.920 – open space, office worker	0.253

348

Table 4. (Continued)

Parameter name	DOE value	*RAC* value
Precipitation rate	0.381 m y^{-1}	0.381 m y^{-1}
Irrigation rate	1.0 m y^{-1} – residential 0 m y^{-1} – open space, office worker	1.0 m y^{-1}
Irrigation mode	Overhead	Overhead
Runoff coefficient	0.004	0.004
Watershed area	8,280,000 m^2	8,280,000 m^2
Accuracy for water/soil computations	0.001	0.001
Density of uncontaminated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Uncontaminated zone total porosity	0.3	0.3
Uncontaminated zone effective porosity	0.1	0.1
Uncontaminated zone hydraulic conductivity	44.5	44.5
Uncontaminated zone b parameter	10.4	10.4
Density of saturated zone	1.8 g cm^{-3}	1.8 g cm^{-3}
Saturated zone total porosity	0.3	0.3
Saturated zone effective porosity	0.1	0.1
Saturated zone hydraulic conductivity	44.5	44.5
Saturated zone hydraulic gradient	0.15	0.15
Water table drop rate	0 m y^{-1}	0 m y^{-1}
Well pump intake depth	10 m	10 m
Nondispersion/mass balance	Nondispersion	Nondispersion
Well pumping rate	250 m^3 y^{-1}	250 m^3 y^{-1}
Thickness of uncontaminated, unsaturated zone	3 m	3 m
Length parallel to aquifer flow	200 m	200 m
Elapsed time of waste placement	0 y	0 y
Dilution length	3 m	Not required for RESRAD Version 5.82
Shape factor	Circular	Based on results of soil concentration analysis
Plant food, contamination fraction	1.0	1.0
Drinking water, contamination fraction	Not used	1.0
Mass loading for foliar deposition	0.0001 g m^{-3}	0.0001 g m^{-3}
Depth of roots	0.9 m	0.9 m
Groundwater fractional usage, irrigation	1.0	1.0
Average storage time for fruits, nonleafy vegetables, and grain consumption	14 d	14 d
Average storage time for leafy vegetable consumption	1 d	1 d
Average storage time for well water and surface water use	1 d	1 d

The Groundwater/Drinking Water Pathway

Groundwater is an extremely complex pathway (described in Task 2), and *RAC* will not assess it in significant detail in the soil action level project because of the extensive ongoing research and the complexity of the interacting processes. We will, however, provide bounding level, screening calculations for the rancher-based scenarios with contaminated drinking water as a pathway for dose. The intent of doing this is not to provide quantitative results but rather to assess the potential importance of the drinking water pathway and provide a mechanism for making calculations when groundwater parameters have been more accurately determined.

For the drinking water pathway, as it will be used in these calculations, the contaminated fraction of drinking water is 1.0; that is, 100% of the receptors' drinking water comes from contaminated groundwater and is as contaminated as the groundwater. By setting the drinking water contamination equal to that in the groundwater, we protect receptors from groundwater resources near their source, thus, protecting the resource at farther downgradient locations.

To explore the sensitivity of the drinking water pathway, we used a deterministic calculation of dose. The parameter values for the five sensitive parameters identified above were not changed from those used in the previous analysis (DOE/EPA/CDPHE 1996) for this sample calculation. For the remaining parameters, we used the values defined in Table 4, the scenario parameters associated with the previous analysis' hypothetical resident, the initial concentration ratios defined in Table 2, and an initial concentration of ^{239}Pu of 500 pCi g^{-1} (18,500 Bq kg^{-1}). This definition of initial concentrations is important in this analysis because we will use dose as the endpoint for comparison.

The maximum annual dose from all radionuclides calculated without the inclusion of the drinking water pathway was 29 mrem y^{-1} (0.29 Sv y^{-1}) at time t = 0. The maximum dose, including the drinking water pathway, was 117 mrem y^{-1} (1.17 Sv y^{-1}) at time t = 221 years. This dose is primarily from drinking water ingestion.

The increase in dose when the drinking water pathway is included is significant. It is important to understand several things about this calculation. First, the increase in dose was due almost entirely to dose from ^{241}Am as it reached the groundwater. The amount of time it took for the americium to reach the groundwater was dependent on the numerical value of the soil-water equilibrium distribution coefficient, which describes the partitioning of contaminants between solid and aqueous phase. This parameter value is critical when the groundwater model is a simple linear model, as it is in RESRAD. If the value of the distribution coefficient is greater than about 220 cm^3 g^{-1}, the nuclide will not reach the groundwater during the 1000-year RESRAD simulation. Based on the RESRAD conceptual model for subsurface transport and the hydrologic transport parameter used in the simulation, it takes over 200 years for significant concentrations of the americium to reach the groundwater and be available in the drinking water, using the DOE distribution coefficient value of 76 cm^3 g^{-1} for ^{241}Am. This calculation was completed only for illustrative purposes, to demonstrate the potential importance of the groundwater pathway. Distribution coefficient is revisited later in this report.

However, much is unknown about the mechanisms by which americium and other radionuclides are transported through the soil column and into the aquifer. There is an additional degree of uncertainty about the properties of the aquifer. Studies on the mobility of radionuclides in the Rocky Flats environment do reveal some important information. Both plutonium and americium are strongly adsorbed, limiting their mobility considerably. The distribution

350

coefficients indicated by research are quite high for both americium and plutonium at Rocky Flats, indicating a high affinity for the solid phase. Parameters that describe the distribution coefficient, bulk hydrologic properties of the subsurface, and precipitation and infiltration in RESRAD dictate the rate at which radionuclides are transported into the aquifer and, therefore, control the calculation of dose from the drinking water pathway.

The vertical distribution of radionuclides in soil is another indicator of mobility, and this has been described by a number of researchers. Some convincing evidence comes from Webb (1996), which revisited the Rocky Flats study documented in Little (1976) and found that the vertical distribution of plutonium and americium has remained nearly the same over the last 20 years. This vertical distribution decreases with depth in the soil column.

There is, however, a recognized potential for transport of radionuclides attached to small colloid-sized particles. Attachment and subsequent transport of these particles would significantly enhance mobility because they do not behave as a dissolved phase species in terms of their sorption-desorption properties. DOE qualitatively looked at the possibility of this transport in their Resource Conservation and Recovery Act facility investigation/remedial investigation Operable Unit-2 (OU-2) document (DOE 1995a). In the DOE report, a study by Penrose et al. (1990) was cited. The Penrose study suggested that small colloids (<0.45 µm) could transport plutonium and americium over large distances in the subsurface. However, colloids larger than 0.45 µm are basically immobile under the same conditions that made small colloid transport possible. Analytical groundwater data from OU-2 for filtered (with a 0.45-µm filter) and unfiltered samples were compared. These data suggested that most of the plutonium and americium in groundwater was associated with the unfiltered sample and, therefore, with particles larger than 0.45 µm in diameter. This qualitative analysis seems to indicate that colloidal transport is not a mechanism by which significant quantities of plutonium and americium are transported to the groundwater at Rocky Flats.

Other studies suggest the opposite is true. Kersting et al. (1999) looked at the possibility for colloidal transport in groundwater at the Nevada Test Site. The researchers observed that radionuclide concentrations in groundwater were associated with the colloidal fraction, and they showed the plutonium source to be an underground nuclear test site 1.3 km away from the groundwater well.

Honeyman (1999) agreed that colloidal transport was certainly a potential and probable mechanism for radionuclide transport, but it pointed out the flaws in the Kersting study. Honeyman recited the three conditions that must be met for colloidal transport to be defensibly proved: (1) colloids must be present in the groundwater, (2) contaminants must associate with the colloids, and (3) the combination of the colloid and contaminant must move through the aquifer. Kersting et al. proved only the first two of these three conditions to be true in their study. In fact, Kersting et al. pointed out the possibility that the study conditions (i.e., increased well pumping) may have enhanced colloidal concentration, preventing quantification of the colloidal load.

The importance of the above discussion is to point out that, at the present time, very little is understood about the mechanisms of colloidal transport of radionuclides in groundwater aquifers. Evidence seems to show that this transport mechanism may be important, but this is an area of current research. Applying any detailed model requires field investigations of the site hydrology and a modeling effort that spans several years to calibrate model results with field measurements.

We looked at the significance of the groundwater/drinking water pathway in this document in terms only of its *potential* for dose. Any dose values resulting from drinking water pathway calculations cannot be finalized during the course of this project simply because the pathway is far more complex than its representation in RESRAD and neither the transport properties nor the aquifer properties are understood at Rocky Flats.

What we learned from this analysis is that groundwater can have an impact on dose that needs to be recognized. Because of the severe limitations on time and resources in this study, we can only recommend that a future study be directed toward this type of work, particularly looking at the migration of ^{241}Am and its progeny.

352

UNCERTAINTY DISTRIBUTIONS

In this project, the term uncertainty usually implies lack of knowledge about the value of a model parameter or the accuracy of a model prediction. We represent these uncertainties as probability distributions. This lack of knowledge about a parameter value can arise from (a) variability of the parameter over space or time, (b) variability among different experiments or field studies that measure the parameter, or (c) variability within individual studies in which measurements, by design, are taken under different sets of controlled conditions. If the data available to us correspond to times, locations, or conditions other than those relevant to this study, then the variability within our limited data (expressed, for example, by the sample standard deviation) may not adequately reflect the uncertainty of the estimates.

Some environmental parameters are difficult to observe directly, and estimates must be based on inferences from available observations of other presumably correlated quantities. But such an indirect approach usually relies on a model connecting the desired quantity with the ones being measured, and use of the idealized model usually introduces uncertainties of its own. An example relevant to Rocky Flats is resuspension. Factors for wind-driven resuspension have been calculated as the ratio of the air concentration of a contaminant (e.g., becquerel of plutonium per cubic meter) divided by the amount of contaminant per square meter of soil (the soil measurement is taken to a depth that is considered resuspendable). A resuspension factor (per meter) is multiplied by a measured soil concentration of a contaminant (e.g., becquerels per square meter) to predict an airborne concentration of the contaminant (becquerels per cubic meter). The implied model assumes a large source area of soil that is uniformly contaminated and uniform in those properties that affect the mechanisms of resuspension (e.g., ground cover, soil particle size distributions, moisture, depth of the resuspendable layer, and terrain topography). It is also assumed that the resuspension factor represents airborne concentrations that are averaged over a sufficient period to be characteristic of the local meteorological conditions. Such uniformities are seldom available to field studies (or applications), and measurements of factors for wind-driven resuspension range from 10^{-4} to 10^{-11} m^{-1} (Sehmel 1972). Without other information, this range is an indication of uncertainty for the local resuspension factor. The resuspension factor for a contaminated location also changes over time as the contaminant migrates downward into soil or undergoes superficial erosion. Anspaugh et al. (1975) and others have made generic characterizations of this temporal trend for plutonium resuspension factors.

Even if direct measurements of the desired quantity are available, they may have been made at a time other than the one relevant to the application. For example, meteorological predictions for environmental assessments often use a joint frequency table of wind speed, wind direction, and atmospheric stability based on five consecutive years of hourly observations at a given location. But when the time of interest for predictions is not within the 5-year period, use of this frequency table introduces a component of uncertainty that results from the variability of the meteorological frequencies over time. This component can be as much as a factor of 2 in predicted annual-average air concentrations, and it is not the only component of uncertainty in such predictions.

In this report, we propose distributions of uncertainty for various parameters that are inputs to RESRAD. To make predictions that reflect these uncertainties, we sampled values for the affected set of RESRAD parameters from these probability distributions, ran RESRAD to calculate the outcome, stored the outcome, and repeated the cycle many times, sampling from the

353

assumed distributions each time. The set of results forms a distribution of outcomes that represents the propagated parameter uncertainties. This distribution might represent dose, dose-to-source ratio, or soil concentration/action level.

The parameters emerging from the sensitivity analysis as important for these calculations were area of contaminated zone, distribution coefficient, mass loading, and mean annual wind speed. As the most critical parameters, it was important to develop distributions of values, where appropriate, using a combination of site-specific data and information from the open literature. This section describes the treatment of these parameters for the independent calculation.

Distribution Coefficient

The transport of radionuclides in groundwater involves solving two fundamental equations that describe a) movement of water within the geologic media, and b) movement of the dissolved constituents (radionuclides). Movement of water is typically described by quantifying water fluxes and velocities (which are functions of the hydrologic properties of the system and the level of saturation) in the system and must be determined first before proceeding with the contaminant transport calculations. Movement of water in porous media, particularly in unsaturated and fractured media, is an area of ongoing research, and much of the overall uncertainty related to groundwater models can be attributed to lack of understanding and poor characterization of these processes. Assuming these processes have been adequately characterized, we then apply the contaminant transport equations to calculate concentrations of radionuclides in pore water at a selected receptor location. Most radionuclides form two phases in groundwater; a dissolved phase that travels with the water, and a sorbed phase that remains attached to the porous matrix. The degree at which a radionuclide sorbs depends on the chemistry of the pore water, the porous media, and the radionuclide itself. At relatively dilute concentrations, the ratio of the concentration in the attached or sorbed phase to that in the pore water remains constant at equilibrium. This ratio defines the linear sorption or distribution coefficient (K_d) and is given by

$$K_d = \frac{C_s}{C_w} \tag{3}$$

where
C_s = the concentration of radionuclide sorbed onto the porous matrix (Ci g^{-1})
C_w = the concentration of the radionuclide in the pore water (Ci mL^{-1})

The distribution coefficient relationship is assumed to be valid over the ranges of concentrations encountered in the environment. In addition, sorption reactions are assumed to occur quickly and achieve equilibrium conditions over the time spans considered (1 to 1000 years). In reality, the sorption process is much more complicated than suggested by the simple distribution coefficient, and is an area of ongoing research. Much of the uncertainty associated with groundwater transport calculations may be attributed to the simplistic treatment of sorption processes. However, without substantially greater resources and time, there is little we can do but resign ourselves to using the distribution coefficient approach in our simulations.

Sorption reactions have the net effect of slowing down or retarding the movement of radionuclides in groundwater. The higher the distribution coefficient, the higher the degree of sorption and the slower the contaminant moves in groundwater. If the radionuclide is non-

354

reactive, that is, it does not sorb and remains entirely in the aqueous phase, its average velocity in groundwater is the same as the water.

Values for K_d vary greatly with physical and chemical properties of the solid, liquid, and radionuclide. Distribution coefficients tend to be greater for finer-grained materials such as silt and clay compared to coarser materials like sand or fractured igneous rocks because the finer materials have cation-exchange capacity. Generally constant for a system under specified conditions, the value for K_d can range over orders of magnitude for different situations, and many of these different situations may exist in the strata of different geologic properties that underlie a given aboveground area. Consequently, the K_d tends to be one of the more sensitive parameters in any calculation involving groundwater.

Values for K_d have been predicted for plutonium, uranium, and americium in the environment around the 903 and Mound Areas (DOE 1995a). The Actinide Migration Studies Panel initiated measurements of K_d in a limited portion of the Rocky Flats environment for uranium and plutonium. Distribution coefficients have also been reported in the literature for a variety of environments. We used all this information to derive a probability distribution for K_d values in the Rocky Flats environment.

The values for the K_d used in the DOE/EPA/CDPHE soil action level document were derived from data reported by Dames and Moore (1984). Dames and Moore reported a range of values for the retardation factor from the literature. The retardation factor is derived from the contaminant mass balance in porous media.

$$C_T = C_w \theta_w + K_d C_w (1 - (\theta_w + \theta_a)) \rho_s \tag{4}$$

where
C_T = radionuclide concentration in the porous media (Ci mL^{-1})
C_w = radionuclide concentration in the water phase (Ci mL^{-1})
θ_w = water filled porosity
θ_a = air filled porosity
ρ_s = particle density (g mL^{-1})
K_d = distribution coefficient (mL g^{-1})

Assuming the total porosity is equivalent to the effective porosity, and relating the bulk density (ρ_b) to the particle density [$\rho_b = \rho_s(1-(\theta_a + \theta_w))$], Equation 4 can be solved for C_w giving

$$C_w = \frac{C_T}{\theta_w \left(1 + \frac{K_d \rho_b}{\theta_w}\right)} \tag{5}$$

The term, $1+K_d\, \rho_b/\theta_w$ represents the retardation factor (R). Solving to K_d yields

$$K_d = \frac{(R-1)\theta_w}{\rho_b} \quad (6)$$

where
K_d = distribution coefficient (cm^3 g^{-1})
R = retardation factor
θ_w = effective porosity of the aquifer
ρ_b = bulk soil density (g cm^{-3}).

For the DOE determination of K_d, the values for θ_w and ρ_b were 0.10 and 1.84 g cm^{-3}, respectively. These values were measured for OU-2 and represent a reasonable estimate of site-specific parameters (DOE 1995a). The Dames and Moore (1984) retardation factor values for sand and clay soils are shown in Table 5 for each radionuclide, along with the associated K_d value calculated using equation (6).

Table 5. Dames and Moore (1984) Reported Retardation Factor Values and Calculated Distribution Coefficients

	Sand		Clay	
Radionuclide	R	K_d (cm^3 g^{-1})[a]	R	K_d (cm^3 g^{-1})
Americium	300	16.3	2500	136
Plutonium	840	45.6	7200	391
Uranium	840	45.6	7200	391

[a] The use of the K_d units of cm^3 g^{-1} are RESRAD driven. These units are equivalent to L kg^{-1}

DOE/EPA/CDPHE (1996) used the midpoint of the ranges shown in Table 5 for americium and plutonium to represent the K_d values for their calculations. Sheppard and Thibault (1990) reviewed a number of distribution coefficient measurements and produced ranges of K_d values for sand, loam, and clay soils. These ranges are shown in Table 6.

Table 6. Ranges of Distribution Coefficients from Sheppard and Thibault (1990) (in units of cm^3 g^{-1} or L kg^{-1})

	Sand		Loam		Clay	
Radionuclide	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
Americium	8.2	300,000	400	48,309	25	400,000
Plutonium	27	36,000	100	5,933	316	190,000
Uranium	0.03	2,200	0.2	4,500	46	395,100

Till and Meyer (1983) reported values for K_d for a variety of nuclides, but of the radionuclides of interest to this study, they showed values only for uranium. Table 7 shows the range of K_d values reported in their work.

350

Table 7. Range of K_d for Uranium Reported in Till and Meyer (1983)

Type of soil, uranium oxidation, and pH	K_d ($cm^3\ g^{-1}$ or $L\ kg^{-1}$)
Silt loam, U(VI), Ca-saturated, pH 6.5	62,000
Clay soil, U(VI), 5mM $Ca(NO_3)_2$, pH 6.5	4400
Clay soil, 1 ppm UO^{+2}, pH 5.5	300
Clay soil, 1 ppm UO^{+2}, pH 10	2000
Clay soil, 1 ppm UO^{+2}, pH 12	270
Dolomite, 100-325 mesh, brine, pH 6.9	4.5
Limestone, 100-170 mesh, brine, pH 6.9	2.9

The Actinide Migration Studies were established to specifically study different aspects of actinide migration and transport in the Rocky Flats environment. In a paper submitted to the panel, Honeyman and Santschi (1997), values of K_d for uranium and plutonium were reported. The authors cautioned that the data presented in their paper represented an upper range of likely values and that another study to determine the lower range of likely values needed to be completed.

Plutonium K_d values were measured in 903 Area lip soils. Uranium values were measured only for the oxidation state U(VI). Uranium geochemistry reveals that the U(VI) oxidation state is the most stable of the three most common oxidation states (U[IV], U[V], and U[VI]) and would also be the most mobile of these three states. Uranium K_d values were measured in solar pond core sediments. Table 8 presents the range of values measured.

Table 8. Range of Maximum K_d Values ($cm^3\ g^{-1}$ or $L\ kg^{-1}$) Measured at Rocky Flats by Honeyman and Santschi (1997)

Radionuclide	Range of possible maximum values
U(VI)	31.2–171
$^{239,240}Pu$	0.98×10^4–1.16×10^5

More than a factor of 5 difference exists between the values measured for uranium, and an order of magnitude exists between the values measured for plutonium. Again, these ranges reflect likely maximum values for K_d. The other data presented here show even larger ranges of values of K_d, which probably more accurately reflect the range of total possible values.

From the data presented in Tables 5-8, it is obvious that K_d values are highly variable and tend to be higher for finer-grained material (clay and silt) compared to coarser grained sands. Also, plutonium and americium K_d values tend to be higher than those for uranium.

RAC has created a distribution of K_d values for uranium, plutonium, and americium. These distributions of K_d values reflect the wide range of variability possible in K_d, giving careful consideration to the Honeyman and Santschi (1997) data set, indicating potential maximum values for K_d in the Rocky Flats system. Although data from Till and Meyer (1983) presented in Table 7 show much higher values of K_d for U(VI) than measured by Honeyman and Santschi, the Honeyman and Santschi data were measured under site-specific conditions. For the purposes of this study, the Honeyman and Santschi conditions were assumed to be representative of Rocky Flats as a whole, and were used to define the upper bound of the K_d distribution for uranium.

For the remaining radionuclides, plutonium and americium, we used the entire range of available data on K_d to define the distribution. The Honeyman and Santschi upper bound for plutonium K_d matches closely with the upper bound reported across the literature, so it was reasonable to use the upper bound reported in the literature. Using the lower bounds identified in the cited literature for all radionuclides allows for the possibility of rapid transport of radionuclides into the groundwater and might help simulate conditions, such as colloidal movement and the special geochemical conditions that promote it, that we are otherwise unable to model.

The distributions were assumed to be lognormal, and the minimum and maximum values described here were assigned to the 0.5% and 99.5% values in the distribution. The properties of a lognormal curve were then used in combination with the two values on the distribution to identify the geometric mean and geometric standard deviation of the distribution. The parameters of the distributions are shown in Table 9.

Table 9. Distributions of K_d Developed for the Independent Calculation (in units of $cm^3\ g^{-1}$ or $L\ kg^{-1}$)

Radionuclide	Geometric mean	Geometric standard deviation
Americium	1800	8.1
Plutonium	2300	5.6
Uranium	2.3	5.4

These distributions of K_d will be used in the independent calculation of soil action levels for Task 5.

It is important to recognize the sensitivity of the K_d value to the aqueous phase concentration and the contaminant transit time. Using ^{239}Pu as an example, we calculated maximum concentration at the receptor well and the time of maximum concentration as a function of the K_d value for a 1 Ci inventory in the source (Figure 1). Maximum concentrations were normalized to the maximum concentration calculated using a K_d value of 2300 mL g^{-1} (1×10^{-9} Ci m^{-3}). The normalized maximum concentration curve ranges over 10 orders of magnitude. The slope of the curve is approximately linear for K_d values less than 1000 mL g^{-1}. For K_d values >1000 mL g^{-1}, the slope increases substantially. The increase in the slope is due to decay effects, because for K_d values >1000 mL g^{-1}, the transit time is greater than the half-life for ^{239}Pu. Also note that for K_d values greater than 100 mL g^{-1}, the time of maximum concentration exceeds 1000 years. Therefore, unless the sampled K_d value is less than 100, groundwater will not be an issue for the scenarios.

358

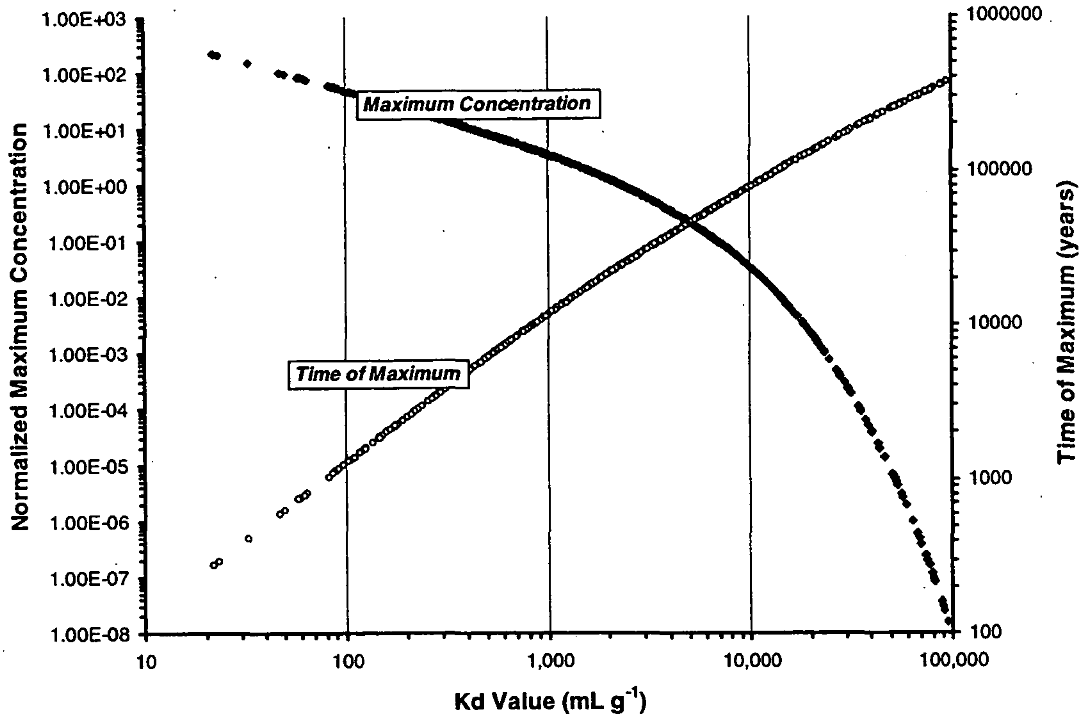

Figure 1. Normalized maximum concentration and time of maximum concentration as a function of the K_d value for ^{239}Pu. Maximum concentrations were normalized to the maximum concentration for a K_d of 2300 mL g^{-1}, which is the geometric mean of the distribution used in the analysis.

We should mention here that the groundwater model employed in RESRAD only considers dissolved phase transport of radionuclides. Recent work by Litaor has suggested that under saturated soil conditions, plutonium can migrate very rapidly. This work is currently unpublished; however, it suggests that certain discrete events, such as heavy rainfall may have moved plutonium into the subsurface in a relatively short period of time. The mechanisms suspected to have resulted in such movement include colloidal transport of plutonium particles through microfractures in the surface soil, and redox reactions coupled with phase changes as a result of saturated conditions near the surface, that temporarily increased the solubility of plutonium. These processes are believed to only operate during periods of heavy rainfall and saturated soil conditions. These processes are still under investigation and are not included in the model, nor can they be given the budget and time constraints of this project. We therefore cannot rule out the possibility that aqueous phase concentrations of plutonium may be underestimated using the approach stated earlier. However, it is our intention to account for the possibility of increased transport conditions such as these by including the lowest measured K_d values available in the literature.

Area of Contaminated Zone

Contamination in soil at Rocky Flats is not uniformly distributed across the site. A number of historic studies have measured concentrations and spatial variation of radionuclides in soil. We used these studies to compile a composite database of soil concentrations at different distances from a significant source of contamination at the site, the 903 Area.

A complication in using RESRAD at Rocky Flats is the highly inhomogeneous spatial distribution of plutonium in the soil. RESRAD works with a specified region of contamination within which the soil concentration is mathematically treated as being uniform, although the developers relax that assumption to accept variation within a factor of 3. Outside the homogeneous region, contamination is assumed to be no greater than background. However, at Rocky Flats, plutonium concentrations in the soil increase by more than a factor of 100 from Indiana Street westward to the 903 Area. Thus, it is difficult to assign a region to a scenario that meets the developers' guidance. If the assigned region is too small, it excludes most of the radioactivity. If it is too large, it fails the test for homogeneity.

To avoid having to conform to RESRAD's definition of contaminated area, we used site data (including air monitoring) to establish relationships between concentrations in air and soil and used these relationships in applying RESRAD to the site. To carry out this task, it was necessary to construct a model of ^{239}Pu concentration in soil as a function of location.

To develop this model, we began with a suitable database of observations. We restricted our selection, for the most part, to measurements for which the documentation included the sampling depth and an approximate time when the samples were taken. One series of measurements that did not meet these criteria is discussed below. The sampling depth is important because recent field and theoretical work reported by Webb et al. (1997) established a fractional concentration depth profile for ^{239}Pu at Rocky Flats that can be applied generically to adjust samples taken at various depths to a common basis.

In general, we followed the example of Webb et al. (1997) and used the ^{239}Pu concentration in the 0–3-cm (0–0.03-m) layer as representative of resuspendable soil and plutonium. The generic profile indicated that essentially all plutonium in the soil at Rocky Flats is currently confined to a depth of 20 cm (0.2 m), with a concentration that decreases with increasing depth. We then adjusted concentrations based on samples taken to depths <20 cm to the 0–3-cm depth by hypothesizing a profile for the sample that was proportional to the standard of Webb et al. (1997). The calculation accounted for plutonium that might have migrated beyond sampling depths less than 20 cm, and a consistent proportion was assigned to the 0–3-cm layer.

Evolution of the depth profile over time is less clear. It appears that after its windborne transport from the 903 Area, plutonium migrated within a few years (at most) into the soil where it was deposited and established the 20-cm profile. Krey and Hardy (1970) indicated that plutonium had already migrated beyond the 13-cm depth. Poet and Martell (1972) questioned this conclusion, reporting that most of the plutonium at seven sites they had sampled was confined to the 0–1-cm layer. They asserted that most of the plutonium found at greater depths in the Krey and Hardy (1970) study occurred at sites that were remote from the 903 Area and in locations where soil had been disturbed. Krey (1974) subsequently defended the conclusion of Krey and Hardy (1970).

Webb (1996) summarized estimates of the soil plutonium inventory from several investigations. These estimates are consistent with a regression curve that shows an initial

360

removal of about 40% of the inventory from the 0–3-cm layer in 10 years (Figure 2). The term "regression" refers to a statistical procedure that fits a function or model, which might be visualized as a curve, to a set of data. The procedure can be extended to use the distances of the data points from the fitted curve (called "residuals") to estimate uncertainties in quantities associated with the model.

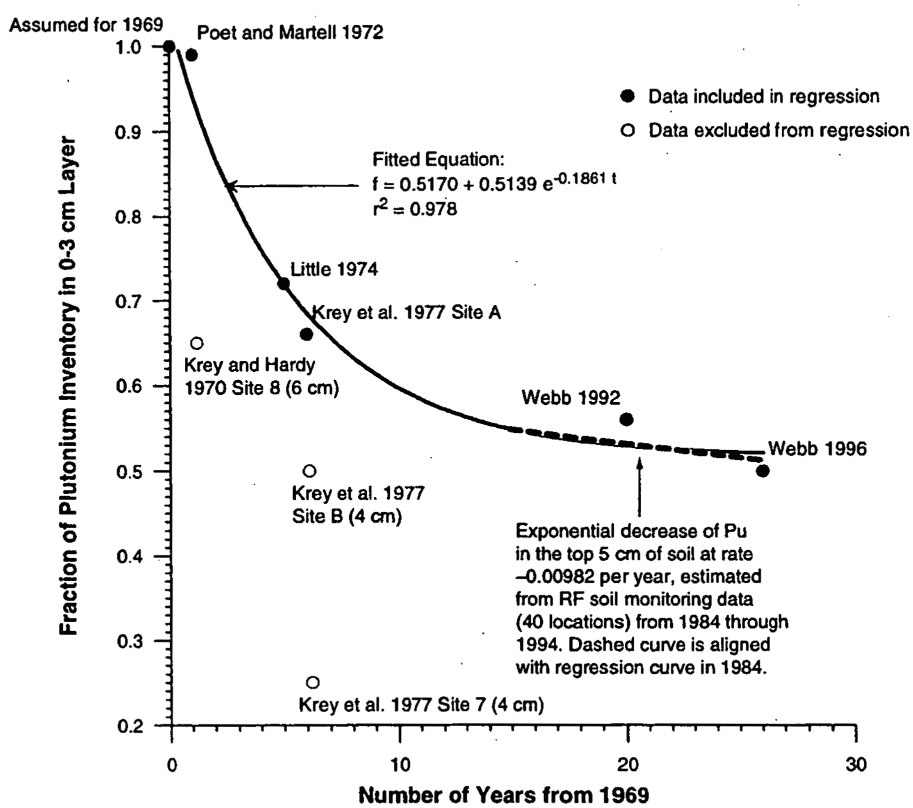

Figure 2. Regression curve fitted to ^{239}Pu data for the 0–3-cm layer of soil at Rocky Flats. The regression was based on data summarized by Webb (1996), which are plotted as black circles. The open circles were excluded from the regression. The dashed line represents an estimated exponential removal rate of plutonium from the 0–5-cm layer measured at 40 stations from 1984 through 1994.

This regression curve, presented in Rood and Grogan (1999), indicates an asymptotic[b] level of about 52% of the initial deposition of plutonium remaining in the 0–3-cm layer. This schedule of decreasing plutonium concentrations is too gradual to be consistent with the conclusions of Krey and Hardy (1970) and with some observations of Krey et al. (1977). Data from some of the locations sampled in these two studies were omitted from the regression because of the apparently inconsistent interpretations. These omitted observations are presented as open circles in Figure 2. Rood and Grogan (1999) has a fuller discussion of the issues involved. The regression curve in

[b] asymptotic refers to the gradual approach of the descending curve to a horizontal line

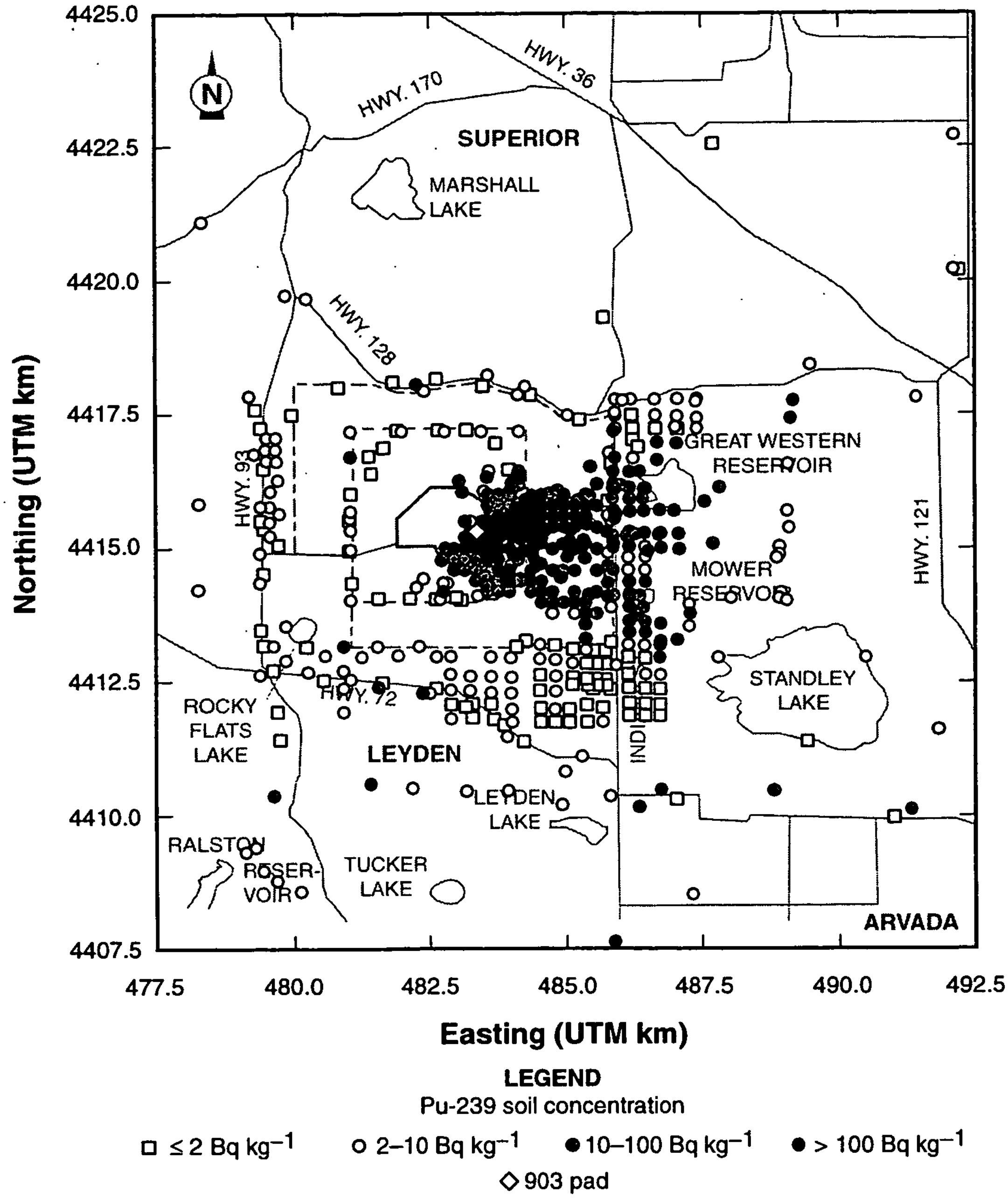

Figure 3. Locations of more than 588 soil samples of ^{239}Pu at Rocky Flats used as a basis for a spatial model (100 Bq kg^{-1} = 2.7 pCi g^{-1}). The plotted symbols give a rough indication of the large-scale variation of the plutonium concentration. Sources of the data were Illsley and Hume (1979), Ripple et al. (1994), and one series from an archive of M.I. Litaor provided by CDPHE.

To be useful, a spatial model of the plutonium concentration in soil must provide estimates for locations not included in the database by means of interpolation. Also, given the considerable spatial variability in the data, the spatial model must provide smoothing. Some efforts have based

362

estimation of contours on kriging methods (Litaor et al. 1995). The *RAC* approach to smoothing was based on the more direct assumption that most of the spatial signal is the result of wind transport of contaminated soil particles from the 903 Area; therefore, a polar[c] representation from this center is reasonable.

Webb et al. (1997) points out that power functions[d] have given satisfactory fits to data along transects from the 903 Area. Figure 4 shows power functions fitted to subsets of the *RAC* database that lie near the 60°, 90°, and 120° transects; the black squares represent the data of Webb et al. (1997), which we included in our model's database. The Webb et al. (1997) data are extensively documented. Therefore, they provide a check on the transformation of the remaining data from heterogeneous sampling efforts to the common basis represented by the profile given by Webb et al. (1997). This adjusted density profile was also used for soil particles of diameter <2 mm. The 2-mm cutoff corresponds to the sieving separation of rocks from soil used in preparing most of the samples. In some of the older samples, however, the rocks were pulverized and re-mixed with the soil (Krey et al. 1976). Figure 3 shows good consistency of the larger database with the data of Webb et al. (1997), but it also emphasizes the scatter of the data, generally to about a factor of about 10 above and below the curve. If the data corresponding to a temporal evolution of the soil profile is adjusted (Task 5), there may be some change in the fit, but it would be difficult at this time to predict the general effect.

[c] The term "polar" means that we represent any location by its distance from a center (or pole) and the angle that a line drawn from the center to that location forms with a specified direction, usually north.

[d] Power functions have the formula $y = f(x) = Ax^b$, where A and b are constants determined from the curve-fitting procedure (this is an example of regression). In this case, y is the concentration of ^{239}Pu in the soil and x is the distance from the 903 Area. The graph of a power function plotted on logarithmic axes is a straight line. Therefore, when data that are plotted relative to logarithmic axes indicate a straight-line trend, one assumes that they are likely to be satisfactorily represented by a power function.

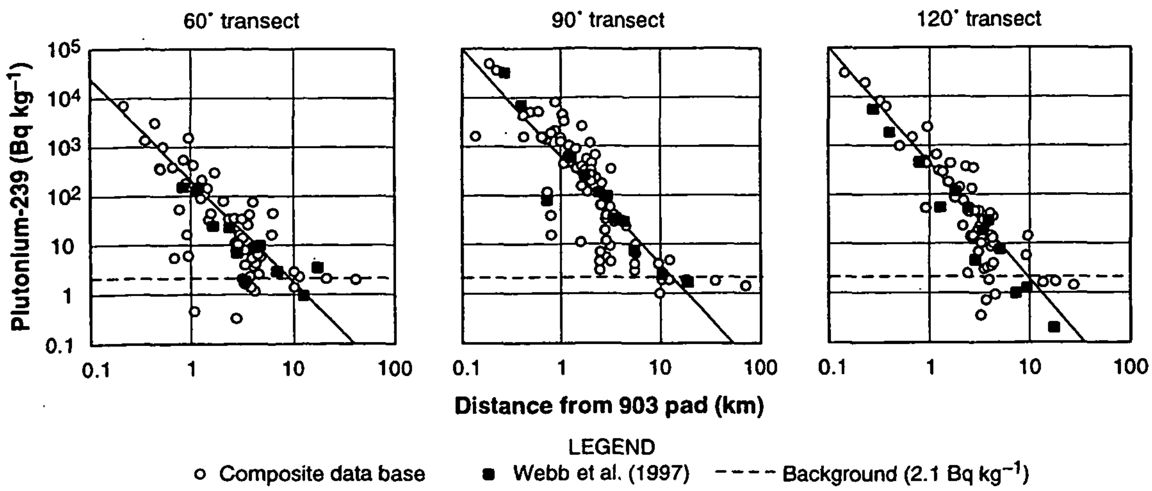

Figure 4. Power function representation of ^{239}Pu concentrations in soil along three transects from the 903 Area. The power functions are straight lines on logarithmic plots. The data of Webb et al. (1997) (black squares) provide a check on the heterogeneous data representing different times and protocols. Data from all sampling depths have been transformed by the profile of Webb et al. (1997) to represent the 0–3-cm layer.

For the spatial soil model, we fitted a power function to the data within each sector of 22.5°, with centerlines at 0°, 22.5°, 45°, etc. To estimate concentration at points on a sector centerline, the model uses the value of the power function from a point near the 903 Area to the distance at which the power function has the value 2.1 Bq kg^{-1}, which is the estimate of background given by Webb et al. (1997). Beyond this distance, all values are assumed to be background for purposes of the model. Between centerlines of sectors, linear interpolation based on the angle is used to estimate the concentration. For two sectors northwest of the 903 Area (292.5° and 315°), the coverage is inadequate to establish credible power function fits, and the power function for 270° was extrapolated to these two sectors. Contours based on the model are shown in Figure 5.

364

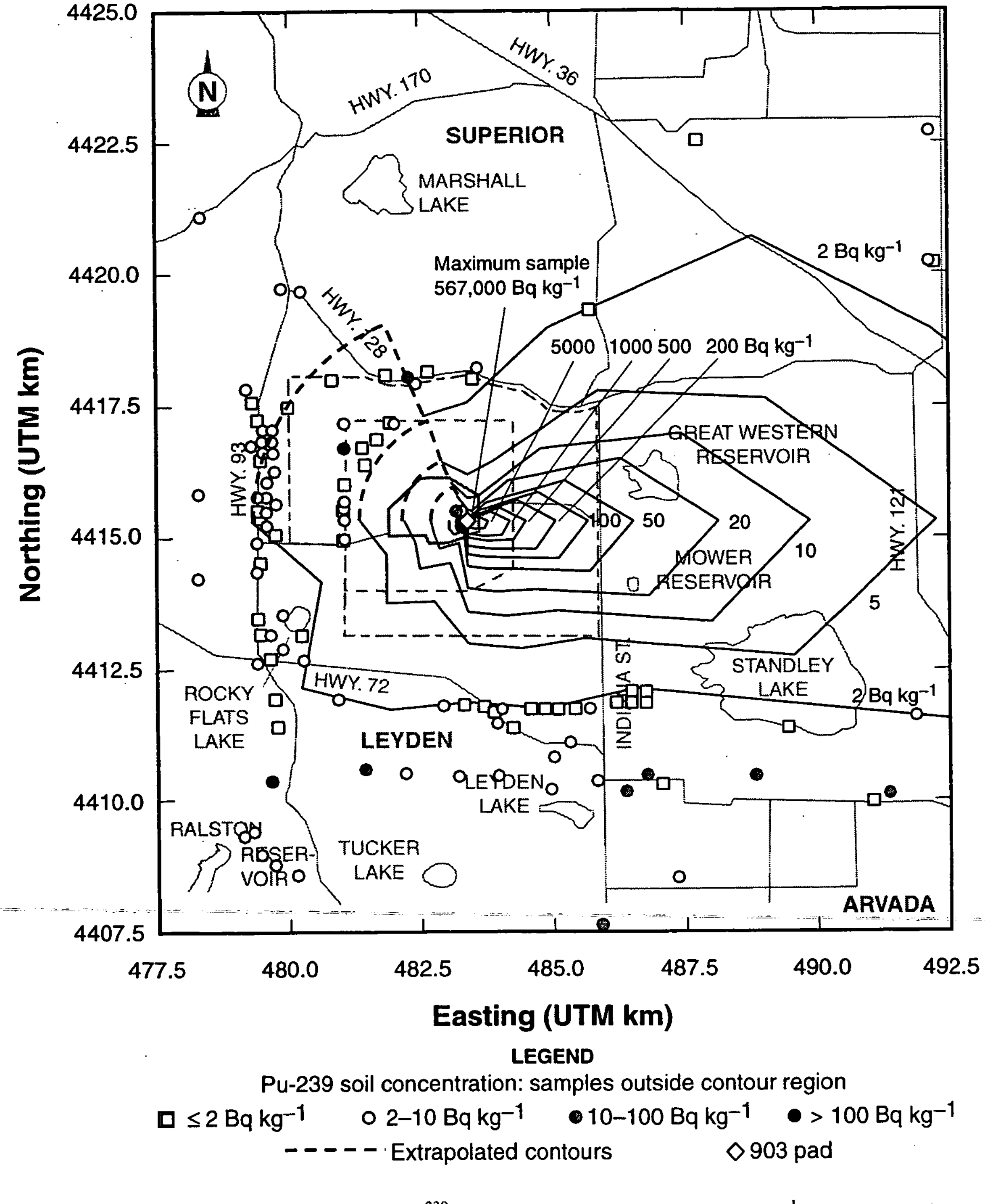

Figure 5. Contours of approximate ^{239}Pu concentration in soil (Bq kg^{-1}) based on the spatial distribution model described in the text.

Dashed lines indicate extrapolation of the two northwest sectors from the 270° sector. Sample locations are shown outside the 2 Bq kg^{-1} (5.4×10^{-2} pCi g^{-1}) contour (approximately background) and within the northwest sectors. Dashed parts of the contours indicate extrapolation where coverage was insufficient for fitting power functions. In these regions and outside the 2 Bq kg^{-1} contour, sample locations were plotted to show that there are some above-background observations where the model would indicate background (2.1 Bq kg^{-1}). However, for purposes of legibility, sample points have been deleted from other regions within the contours.

Although the contours may be considered crude, with an angular resolution no better than the linear interpolation between sectors, they illustrate the considerable variation of the concentrations and the particularly rapid increase as the pad is approached along eastward transects. The model estimates are constrained not to exceed the maximum adjusted sample value (567,000 Bq kg^{-1} or about 15,000 pCi g^{-1}), which occurs in the immediate vicinity of the 903 Area. These contours (or any set of contours based on plutonium concentrations in soil at Rocky Flats) cannot be assumed to provide exact partitions according to magnitude. The smoothing and interpolation provided by the model must be kept in mind. The model is not intended to give accurate estimates at specific locations, but rather it provides a basis for integration[c] of resuspension over large areas for calibration.

Mass Loading Factor

RESRAD bases its calculations of resuspension parameters assuming an area of homogeneous contamination. RESRAD defines an area that has homogeneous contamination as any area where all contamination levels are within a factor of 3 of the mean. An area of higher concentration would then be restricted to an area of no greater than 100 m^2. Figure 5 shows that the soil contamination at Rocky Flats is not homogeneous. Clearly, Rocky Flats soil contamination does not fit within the boundaries of this definition needed for RESRAD.

Because the area of contamination is so closely tied to calculating the resuspension parameter mass loading, we bypassed this calculation in the RESRAD code and had RESRAD estimate resuspension in a different way. The resuspension process, however, is very complex, with a number of mechanisms controlling it that have not been well quantified in spite of the years of research on the topic. Because the best way to evaluate soil resuspension is on a site-specific basis, we calibrated the model to site-specific data.

The RESRAD documentation cautioned that if air concentration values were available for the site under evaluation, these should be used in lieu of the area factor calculation (Chang et al. 1998). There are several sources of air monitoring data across the area of study for the soil action level work. Langer (1991) measured air concentrations at a single location 100 m southeast of the former 903 Area from 1983–1984 and monitored a less instrumented location at the East Gate near the 903 Area. Rocky Flats annual site environmental reports summarize data from several air monitors located throughout the Rocky Flats complex. These monitoring data do not, however, provide particle size information.

The tools that *RAC* used to calibrate resuspension to available air concentration data are described in the Task 5 report. It is important to understand that because of the large degree of inhomogeneity at Rocky Flats, it is difficult to use RESRAD, or most existing assessment

[c] In this context, "integration" may be thought of as adding up the contributions of resuspension arising from many small areas within the contaminated region to estimate their collective effect on air concentration at a single specified location occupied by an air sampler or the subject of a scenario.

366

programs, to make these calculations. Our method provides a way to use the RESRAD tool in combination with available site-specific data to make estimates of resuspension based on actual site conditions.

To make this calibration, *RAC* specified the area that was the domain of an individual receptor. Examples might be a ranch for the rancher, some area of land that the recreational user might cover during an exercise period, or the office buildings and surrounding parking area used by the office worker. In general, this area was a small subregion of the contaminated area. We estimated the variation of the air concentration that existed within the defined domain based on the current state of ground cover, using the existing air concentration data. The resuspension mechanism in RESRAD was then constrained to calculate the estimated air concentration for that receptor. This approach bypassed the generic area factor and resuspension mechanism in RESRAD and defined resuspension based on actual site data.

The calibration of the model used a Gaussian plume air dispersion model to predict the annual averaged contribution to plutonium air concentration at a fixed receptor location from resuspension of contaminated soil. The resuspension rate for the calculation was estimated from a soil concentration given by our soil model, meteorological data, and two parameters that need to be estimated for local conditions. For each wind direction, these computed contributions of resuspended material from small areas were added together to provide a total estimate of air concentration. The results were then averaged over the 16 wind directions, using local meteorological frequencies. A prediction was made for the location of each air sampler with trial values for the resuspension parameters, and the results were compared with the monitoring data. This comparison was used to adjust the resuspension parameters to give the best fit of the predictions to the data. The fitted resuspension parameters provided the calibration.

Using these fitted parameters, *RAC* applied the same integration procedure to estimate the annual average of plutonium air concentration at any location on or near the site. We also estimated plutonium air concentrations based on the assumption of reduced soil concentrations that simulate the results of remediation. The regression also yielded estimates of uncertainty for the predicted air concentrations. These air concentrations enabled us to use RESRAD for calculations of dose and soil action levels for any scenario. Anspaugh et al. (1975) described a similar procedure for estimating resuspension rates using data for plutonium from the Nevada Test Site.

A procedure such as this was required because air concentrations within the domain of a scenario depend not only on soil contamination within that domain, but also on soil contamination throughout a larger upwind region. The extent of this larger region is not well defined.

Krey et al. (1976) reported results of soil and air sampling east of the 903 Area. Their comparison of plutonium activity per gram of airborne dust and plutonium activity per gram of soil led them to the conclusion that only 2.5% of the airborne dust was representative of the soil at the three sites they sampled. The remainder of the airborne dust presumably came from outside the immediate vicinity. An uncharacteristic frequency of rain reported by Krey et al. (1976) during their field work suggests some caution regarding the 2.5% figure.

Table 4.1 of NCRP Report No. 129 (NCRP 1999), however, indicated that 95% of the airborne dust at about a 1-m height comes from an upwind fetch (upwind distance) of 60 m if the ground cover is tall grass (145 m for short grass and 175 m for bare ground). These distances seem too short to be consistent with the observation of Krey et al. (1976). Our calculations suggested that at the locations sampled at Rocky Flats, most of the resuspended dust would have come from onsite. There is literature on the subject of footprints of fluxes (the footprint is the source region for a flux through a specified area, such as a sampler intake). Our method implicitly deals with the question by integrating over a large area that is certain to contain the relevant footprint.

It is important to understand the dependence of this calibration on the current state of ground cover. All the available air monitoring data reflect this ground cover; therefore, any calibration done to these data necessarily includes this assumption. *RAC* has developed resuspension parameters for an extreme situation, such as a fire or other natural disaster, that might remove the grass cover and leave an open soil source available for resuspension.

This calibration was developed as a part of Task 5, *Independent Calculation*. We believe this method will make the best use of RESRAD within its design limits and provide external data for quantities that exceed the design limits for this site. RESRAD is well suited to performing radiological decay chain calculations, concentrations of radionuclides in exposure media (given the concentrations in air that our auxiliary calculation will provide), and annual dose at various future times from multimedia exposure to the radionuclides. The corresponding soil action levels for each scenario depend on the highest plutonium soil concentrations that are consistent with the limiting annual dose for the scenario.

Mean Annual Wind Speed

Mean annual wind speed was not required in the previous version of RESRAD. It is used to calculate the area factor for use in the resuspension calculation in RESRAD Version 5.82. According to the National Climatic Data Center (www.ncdc.noaa.gov), the 43-year annual average wind speed for the Denver area is 4 m s^{-1} (NCDC 1999). This average fluctuates very little for any given year, ranging from about 3.7 to 4.4 m s^{-1}.

As described above, however, *RAC* estimated resuspension by calibrating to site-specific air concentration data. This calibration required the use of wind speed data, but it also used a data set that contains more information than wind speed alone. For example, data on wind direction and atmospheric stability class from the onsite Rocky Flats meteorological station were also included. Further information on the use of the wind speed data will be included in the Task 5 report. The joint frequency tables showing the 5-year average wind speed data for the Rocky Flats area are given in Appendix A to this report. The first six tables in the appendix represent the data for each stability class, with fractional values adding up to 1 for each table. The last table is the composite joint frequency table for all stability classes.

Because there was a recognized potential for high wind events at Rocky Flats, we carefully considered them for this project. During Phase II of the Historical Public Exposure Studies on Rocky Flats, a series of high wind events was predicted to result in a significant quantity of offsite contamination from the 903 Area (Weber et al. 1999). It was demonstrated that these winds resuspended a large amount of the available plutonium from the highly contaminated area.

The largest wind events during the period after the 903 Area barrels were cleared and before the area was covered with asphalt were modeled as six discrete wind events. These events produced the largest degree of dust and contamination suspension from the 903 Area. The high wind events were estimated to have been responsible for most of the activity released from the 903 Area. However, high wind speeds also result in greater dispersion, dilution, and depletion

368

within an airborne plume, resulting in lower air concentrations than would be predicted had the same activity been released over a longer period of time and modeled using annual average meteorological data. This is clear if we consider the plutonium concentrations predicted and reported during the Phase I of the Historical Public Exposures Study on Rocky Flats. Although the source term for respirable particles calculated in Phase I was about the same as that calculated in Phase II, the total integrated concentration value from the Phase I work is a factor of 2 to 3 higher than that from the Phase II work. Consequently, it appears that while the discrete events may have contributed to most of the offsite contamination, they do not appear to be as important from an airborne concentration standpoint.

This is a very important characteristic of high winds. Although at the beginning of the Historical Public Exposures Studies on Rocky Flats, high winds were widely regarded as probably the single greatest contributor to exposure, they were revealed in that study to be responsible instead for reducing the concentrations of contamination in air. As a result, high winds will not be explored further in the soil action level project.

40% of the time. But with a total time onsite of 8760 h y^{-1}, the rancher spends approximately 3500 of those hours outdoors. Each person represented by a scenario is present at the site for a defined period of time.

For the soil action level assessment, the scenarios are described and defined by numerous parameters, some much more important than others. The scenario parameters include breathing rates for various activity levels and ages, soil ingestion rates for children and adults, fraction of time spent indoors and outdoors, and the potential use of or exposure to contaminated water from the area. We have focused our greatest effort on establishing values for breathing rate and soil ingestion, as these are parameters in which the Panel expressed primary interest. For the remaining parameters, we used the literature to select values, which in some cases differ from the RESRAD default values or the DOE/EPA/CDPHE scenarios (DOE/EPA/CDPHE 1996). Table 11 summarizes the key parameter values for all scenarios.

370

Table 11. Scenario Parameter Values for DOE and *RAC* Scenarios

Parameter	DOE/EPA/CDPHE scenarios			*RAC* recommended scenarios			
				Nonrestrictive			Restrictive
	Residential	Open space	Office worker	Resident rancher	Child of rancher (10 y)	Infant of rancher (2 y)	Current site industrial worker
Scenario name	DOE-1	DOE-2	DOE-3	*RAC*-1	*RAC*-2	*RAC*-3	*RAC*-4
Dose limit (mrem y^{-1})	15/85	85	85	15	15	15	85
Onsite location				East of present 903 Area	East of present 903 Area	East of present 903 Area	Present industrial area
Time on the site (h d^{-1})				24	24	24	8.5
Time on the site (d y^{-1})				365	365	365	250
Time on the site (h y^{-1})	8400	125	2000	8760	8760	8760	2100
Time indoors onsite (h y^{-1})				5300	6600	7740	900
Time indoors onsite (%)	100	100	100	60	75	90	40
Time outdoors onsite (h y^{-1})	0	0	0	3500	2100	860	1200
Time outdoors onsite (%)	0	0	0	40	25	10	60
Breathing rate (m^3 y^{-1})	7000	175	1660	10800	8600	1900	3700
Soil ingestion (g y^{-1})	70	2.5	12.5	75	75	75	50
Irrigation water source	Ground-water	NA[a]	NA	Ground-water	Ground-water	Ground-water	NA
Irrigation rate (m y^{-1})	1	NA	NA	1	1	1	NA
Onsite drinking water source	no	no	no	Ground-water	Ground-water	Ground-water	no
Drinking water ingestion (L d^{-1})	NA	NA	NA	2	1.5	1	NA
Drinking water ingestion (L y^{-1})	NA	NA	NA	730	550	365	NA
Fraction of contaminated homegrown produce	1	0	0	1	1	1	0
Fruits, vegetables and and grain consumption (kg y^{-1})	40.1	NA	NA	190	240	200	NA
Meat (kg y^{-1})	NA	NA	NA	95	60	35	NA
Milk (L y^{-1})	NA	NA	NA	110	200	170	NA
Leafy vegetables (kg y^{-1})	2.6	NA	NA	64	42	26	NA

[a] NA = not applicable.

To select appropriate parameters for the scenarios, we reviewed the scientific literature and current EPA and NCRP guidance. For two of the parameters that are particularly important in the

371

scenarios (breathing rate and soil ingestion rate), we fully considered the uncertainty (or variability) distributions of these parameters. For these two parameters, we generated a distribution of values and sampled from the distribution using Monte Carlo techniques. This process considered the available studies equally. The distributions are characterized with a central value, such as the median, and some measure of the spread of the distribution, such as the standard deviation or the 5th and 95th percentiles of the distribution.

In developing a particular scenario and considering variability of a parameter within the population studied, we selected a percentile of the distribution as needed to extend protection to a larger fraction of a potentially exposed population with characteristics similar to those of the scenario subject. After the parameter value was selected from our distribution of values for use in the scenario, the scenario was considered fixed just as standards are fixed as a benchmark against which to measure an uncertain value.

The following sections provide details on selecting the scenario parameters that are expanded or differ from the parameter values given for the current DOE/EPA/CDPHE scenarios.

Breathing Rate

We compiled data from numerous published papers to provide perspective in selecting suitable breathing rates (Table 12). In general, breathing rate studies indicate that gender makes little difference on breathing rates through about age 12. For teens through adulthood, the breathing rate can be 40–50% higher in males than females. There is also age dependency on breathing rates, with adults having breathing rates that are about a factor of 3 higher than for young children. For a person of a given age and gender, the most significant parameter affecting breathing rate is the level of activity; breathing rates can be 15 times higher under maximum work conditions than resting. This activity dependence is important for acute exposure of a few hours, but less important for a continuous chronic exposure of a year.

The time for each *RAC* scenario was divided among three types of activities: sleeping or sedentary, light activity, and heavy activity. For the infant and child, the time was divided into sleeping and light and moderate activities. For the onsite worker, the time was divided between time at the site (hours per day) and time away from the site (hours per day). While at the site, the time spent in light, moderate, and heavy activity was identified. For each scenario, we then assigned duration for the various daily activity levels. The daily breathing rate for each scenario was the time-weighted average breathing rate for each activity level. Although there is no distinction between indoor and outdoor air concentrations in the assessment, the levels for indoor and outdoor activities differed.

Based on published studies, *RAC* created distributions of breathing rates for active and sedentary adults, for active and sedentary children, and for active and sedentary infants. Using these distributions and the recommended breakdowns of daily activity for each receptor, we created distributions of scenario breathing rates for each scenario. *RAC* recommended and the Panel agreed to using the 95th percentile value from these distributions for the scenario breathing rate. Figure 6 shows the distributions for the nonrestrictive scenarios (rancher, child, and infant), and Figure 7 shows the probability distribution for breathing rates for the restrictive scenario (onsite worker).

372

Table 12. Summary of Key Breathing Rate Studies Reviewed

Study	Approach	Group	Breathing rate ($L\ min^{-1}$)
Silverman et al. (1951)	Max inspiration and expiration determined for design of respiratory equipment; one group; adult males	Male athlete,	
		sitting on bicycle	10.2
		heavy exercise	75
Thompson and Robison (1983)	Based on breathing rate at normal body temperature and pressure; nine age groups; infant through adult; male and female	Adult male	
		resting	8.8
		active	30
Roy and Courtay (1991)	Based on time budgets (hours spent at various activities); six age groups; infant through adult; male and female	Adult male resting	7.5
		Adult male, heavy activity	50
Layton (1993)	Based on oxygen uptake associated with energy expenditures and metabolism; seven age groups; infants through adult; male and female	Adult male average	11
		Range based on activity during day	7–12
Finley et al. (1994)	Age-specific distributions for chronic inhalation rates based on Layton (1993)	Adults 50th percentile	8.2
		Adults 5th, 95th percentiles	5.8, 11.6
EPA (1997)	Deterministic;		
	Outdoor workers (15 men; 5 women)	Light activity	12
		Heavy activity	51
	Outdoor construction workers (19 males)	Light activity	24
		Heavy activity	34

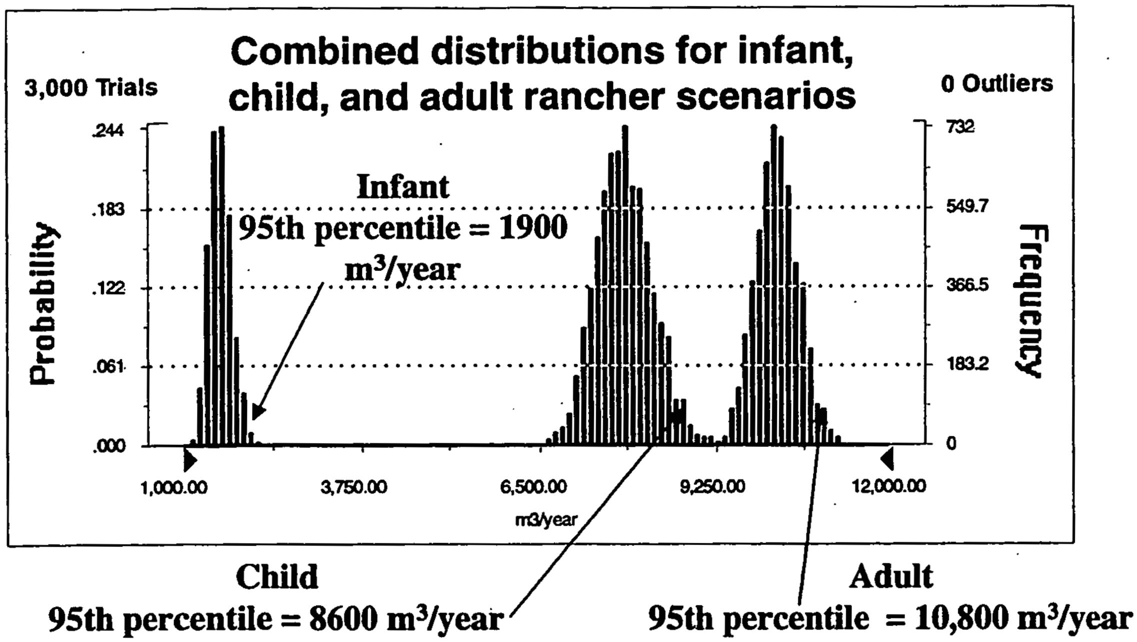

Figure 6. Distributions of breathing rates for the nonrestrictive scenarios: infant, child, and rancher. The 95th percentile of the distribution is shown for each scenario.

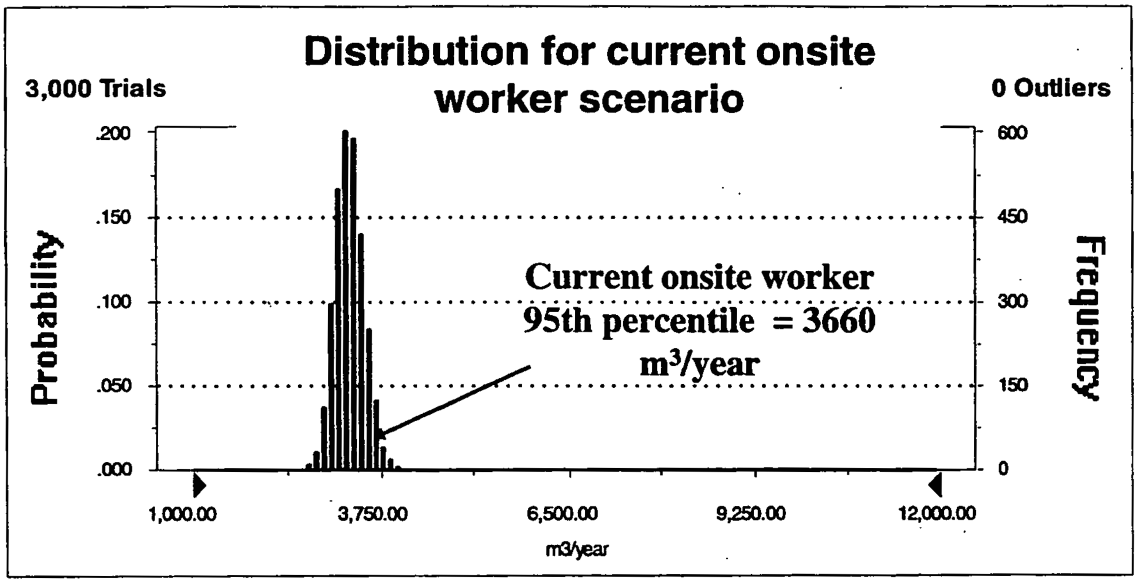

Figure 7. Probability distribution of breathing rate values for the restrictive scenario: current onsite industrial worker scenario. The 95th percentile of the distribution is $3660\ m^3\ y^{-1}$.

374

Soil Ingestion

Various studies have evaluated the unintentional and intentional ingestion of soil by children and adults. Table 13 lists the studies used in selecting the soil ingestion rate for our scenarios. The table summarizes the approach used in assessing ingestion in each study and the geometric mean and geometric standard deviation for those studies. In 1984, the Centers for Disease Control estimated age-specific soil ingestion at about 10 g d^{-1} based on observations of behaviors of children of 1 to 4 years of age (Kimbrough et al. 1984). In 1986, one of the first quantitative assessments of human soil ingestion was carried out using tracer elements in the soil like aluminum, silicon, titanium (Binder et al. 1986). In 1990, Calabrese et al. (1990) studied soil ingestion rates in adults and children using a mass balance approach and more controlled procedures. Simon (1998) developed scenarios based on an extensive review of the literature. The scenarios applicable to this current soil action level study are for a rural lifestyle with homes in a sparsely vegetated area, similar to the Rocky Flats area. Simon assumed a lognormal distribution for inadvertent soil ingestion for adults with a geometric mean of 0.2 g d^{-1} and a geometric standard deviation of 3.2. For children living this lifestyle, the geometric mean is 0.2 g d^{-1}, with a geometric standard deviation of 4.2 to develop a distribution of values, and a median estimate of 0.2 (which would give 5th and 95th percentile values of 0.02 g d^{-1} and 2 g d^{-1}, respectively).

Soil ingestion is difficult to verify and quantify, and some studies do not differentiate between inadvertent or intentional intake. Both inadvertent and intentional soil consumption is seen worldwide, in all cultures, and intentional soil consumption can affect estimates of soil ingestion rates selected for use in this prospective study. During our discussions with the RSALOP, questions arose regarding soil ingestion values and how the extreme behavior of geophasia (intentionally consuming soil) might affect our probability distribution. There was concern that the few geophasic individuals in some of these studies biased our initial use of the 95th percentile value for daily soil ingestion rate extremely high. Many soil ingestion studies have focused primarily on children, leading to a general view that geophasia is more common in young children than other segments of the population. The reason for this conclusion may be that it has been easier to document geophasic children in the more controlled study environments with children. However, there are several studies (e.g., Simon 1998) that cite cases of geophasia in several segments of the population, including adolescents and pregnant women. While this may be more common in indigenous or rural populations, geophasia has been documented in various population subgroups in United States. The incidence of geophasia in the population is quite small, estimated at less than 1%; however, quantitative evaluation of this phenomenon is sparse.

Most studies, even the more recent mass-balance soil ingestion studies (Stanek and Calabrese 1995) are conducted under fairly idealized conditions or during more mild seasons of the year, and authors tend to point this out in their reports (Calabrese et al. 1990; Binder et al. 1986). This timing factor provides conditions where children may have more ready access to open play areas and outdoor activities and adults are more involved in gardening activities. While values derived from studies conducted from a few days to a few weeks are quite valid in estimating daily soil ingestion rates, there is a need to carefully consider the implications of translating this daily soil ingestion rate to an annual soil ingestion rate.

Table 13. Summary of Soil Ingestion Studies Reviewed

Study	Approach	Soil ingestion ($g\ d^{-1}$)	
		Geometric mean	Geometric std dev
Simon (1998) NCRP Report 129 (NCRP 1999)	Scenarios based on literature review: Rural lifestyle (w/homes)—sparsely vegetated		
	Lognormal adults (inadvertent)	0.2	3.2
	Lognormal children (inadvertent)	0.2	4.2
Thompson and Burmaster (1991) (reanalysis of Binder et al. 1986)	Lognormal distribution (children)	0.06	2.8
Stanek and Calabrese (1995) (reanalysis of Calabrese et al. 1990)	Range of median soil ingestion of 64 children over 365 days	0.001–0.10	
	Median of daily average soil ingestion of 64 children:	0.075	
	Range of upper 95% soil ingestion estimates	0.001–5.3	
	Median upper 95% soil ingestion estimate of 64 children over 365 days	0.25	
Calabrese et al. (1990) (children)	Distribution percentiles Median (5th, 95th percentiles)	0.02 (0, 1.2)	
Thompson and Burmaster (1991) (included geophasic children)	Distribution percentiles Median (5th, 95th percentiles)	0.06 (0.01, 9)	
Kimbrough et al. (1984) (children)	Deterministic Mean (low, high)	0.1 (0.05–5)	
Hawley (1985) (adults)	Deterministic (average estimate)	0.06–0.07	
EPA (1997) (adults)	Deterministic (conservative)	0.1	
NCRP Report 123 (NCRP 1996)	Deterministic (conservative)	0.25	

The daily soil ingestion rates are based on a few days or weeks of measurements during times when the soil ingestion may be more likely because of weather conditions or available surface soil. When converting this rate to an annual intake, care must be given because the year includes large periods of time where outdoor inadvertent soil ingestion activities may be somewhat limited by snow cover, frozen ground, and inclement weather. For these reasons, we will use the 50th percentile of our distribution for our daily soil ingestion rate. From the daily soil ingestion rate, we will then calculate an annual soil ingestion value based on the number of days of exposure.

We reviewed various published soil ingestion studies and fit a probability distribution to the data from these studies (NCRP 1999; Simon 1998; Stanek and Calabrese 1995; Thompson and Burmaster 1991; Calabrese et al. 1990). We then looked at how deterministic values from other studies fit into the probability distribution (Kimbrough et al. 1984; EPA 1997; NCRP 1996; Hawley 1985). Figure 8 shows the probability distribution for the soil ingestion studies. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown. As stated above, we used the 50th percentile of this distribution (0.2 g d^{-1}) as the daily soil ingestion rate for our scenarios.

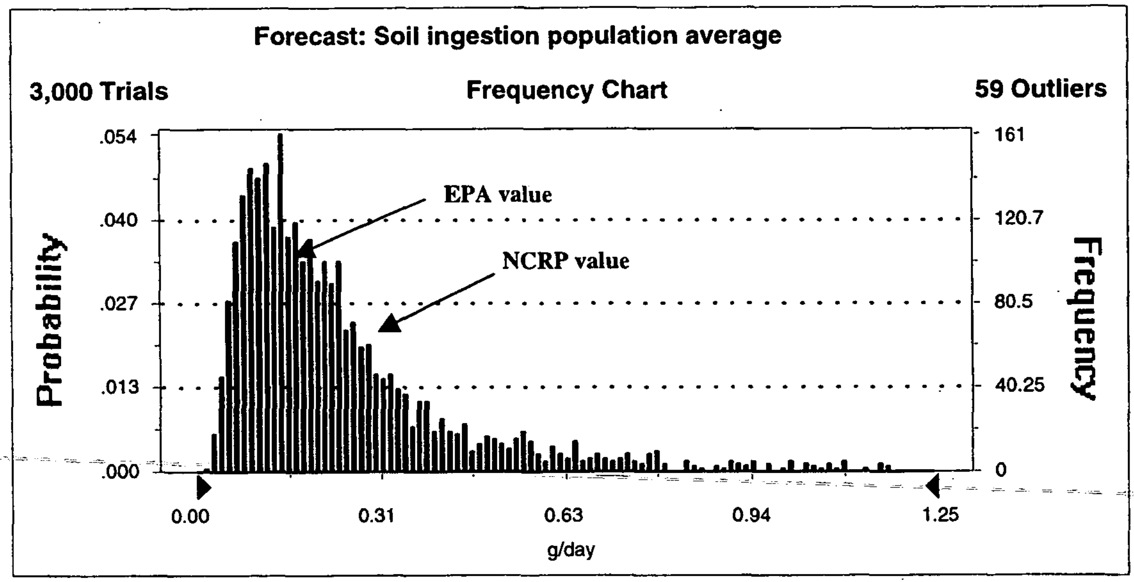

Figure 8. Frequency distribution of soil ingestion values from CrystalBall©. The resulting distribution fits well to a lognormal distribution with the following parameters: median = 0.2 g d^{-1}, the 5th percentile = 0.06 g d^{-1}, and the 95th percentile of 0.73 g d^{-1}. The geometric standard deviation is 2.17. The current EPA value of 0.1 g d^{-1} and the NCRP value of 0.25 g d^{-1} are shown.

378

Groundwater as Irrigation and Drinking Water Source

While groundwater was a source of drinking water and irrigation for the rancher scenario, it has been emphasized that no elaborate calculations can be undertaken for this pathway within the scope of this project. The effort will be restricted to the models and mechanisms that are incorporated within the codes under consideration, with all relevant caution. The irrigation fraction from groundwater for the rancher scenario was 1.0, the RESRAD default value. The contamination fractions of drinking water and irrigation water for the rancher scenario were both 1.0, the default parameter values for RESRAD.

As discussed in the Task 2 report for this project (Killough et al. 1999), the DOE/EPA/CDPHE scenarios (DOE/EPA/CDPHE 1996) did not include the groundwater and surface water pathways because (1) the site streams (Woman and Walnut Creeks) are perennial and would not provide a reliable year-round water source for an individual living on the site and (2) surface aquifers underlying the site do not produce enough water for domestic or agricultural use. The aquatic food pathway was eliminated because the streams are not capable of sustaining a viable fish population. We have reviewed the DOE/EPA/CDPHE approach and agree with their conclusions with regard to surface water pathways. Regarding the groundwater pathway, however, it is not unreasonable to assume for the rancher scenario living under subsistence conditions, a water well that produces 2 gal min^{-1} (DOE 1995b) would be adequate to provide drinking water and perhaps water for a few head of livestock and some limited irrigation. By addressing these pathways, even on a screening level, we can evaluate their potential importance.

Drinking Water Intake

We recommended a drinking water intake of 2 L d^{-1} (730 L y^{-1}) for the adult rancher scenario, 1.4 L d^{-1} (550 L y^{-1}) for the child of the rancher, and 1 L d^{-1} (365 L y^{-1}) for the infant of the adult rancher. These values are based on regulatory guidance from the EPA (1989, 1997) and from other studies (Finley et al. 1994). The current DOE/EPA/CDPHE scenarios did not include drinking water as a potential pathway. The RESRAD default value for drinking water ingestion is 510 L y^{-1}.

Fruits, Vegetables, and Grain Consumption

Annual consumption of major food groups as a function of age for the United States have been estimated and reported by various agencies. This information was necessary in our assessment in order to calculate an average dose from ingestion of produce and grains grown in the contaminated soil, or of meat and milk ingested from animals that ate vegetation grown on the site. In a recent publication, NCRP (1999) compiled values from a number of sources for consumption of major food groups. We recommended an annual consumption rate for fruits, nonleafy vegetables, and grains of 190 kg y^{-1} for the rancher scenario, 240 kg y^{-1} for the child scenario, and 200 kg y^{-1} for the infant scenario (Table 5.1, NCRP 1999). Consumption of leafy vegetables is assessed separately in RESRAD. For the *RAC* scenarios we assumed the consumption of leafy vegetables at the rate of 64 kg y^{-1} for the rancher, 42 kg y^{-1} for the child, and 26 kg y^{-1} for the infant scenarios (Table 5, NCRP 1999). The DOE/EPA/CDPHE scenarios assumed 40.1 kg y^{-1} of vegetables, fruits, and grains and 2.6 kg y^{-1} of leafy vegetables. The

378

RESRAD default values for these parameters are 160 kg y^{-1} for fruits, nonleafy vegetables, and grains and 14 kg y^{-1} for leafy vegetables.

Milk and Meat Consumption

We recommended an annual ingestion rate for milk of 110 L y^{-1} for the adult rancher, 200 L y^{-1} for the child, and 170 L y^{-1} for the infant (NCRP 1999). For meat consumption, we recommended a value of 95 kg y^{-1} for the adult rancher, 60 kg y^{-1} for the child, and 35 kg y^{-1} for the infant (NCRP 1999). These pathways were not assessed for the DOE/EPA/CDPHE calculation.

CONCLUSIONS

To develop meaningful and appropriate calculations of soil action levels at Rocky Flats, *RAC* collected site-specific data and presented them in this report. Data of this type will be used for all parameters that were revealed as sensitive to change and parameters that warranted adaptation based on the information available in the literature. Not every parameter necessary for the use of RESRAD was changed from its value in the original set of calculations (DOE/EPA/CHPHE 1996). Changes were often not necessary because the values were not sensitive to change, and effort expended on these parameters was not warranted. The primary effort in this report was directed toward the most important parameters for soil action level calculations with RESRAD: mass loading, soil-to-plant transfer factors, distribution coefficients, area of contamination, and mean annual wind speed.

Task 5 of this project, *Independent Calculations*, will use the values and distributions presented here in the calibrated version of RESRAD. Values for soil action level and dose will be presented as distributions of possible values for each individual scenario.

380

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382

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384

APPENDIX A:
JOINT FREQUENCY TABLES
FOR FIVE YEAR AVERAGE
ROCKY FLATS WIND SPEEDS

385

Stability Class A

Fraction of total meteorological data in stability class = 0.0133

	Windspeed (m s^-1^) 0.75	2.3	4.1	6.7	9.5	11	Fractional Direction Totals	Avg. windspeed (m s^-1^)
N	0	0.01504	0	0	0	0	0.01504	2.3
NNE	0	0.04511	0	0	0	0	0.04511	2.3
NE	0	0.09774	0	0	0	0	0.09774	2.3
ENE	0.00752	0.18045	0	0	0	0	0.18797	2.23799
E	0.01504	0.18045	0	0	0	0	0.19549	2.18075
ESE	0	0.18045	0	0	0	0	0.18045	2.3
SE	0	0.10526	0	0	0	0	0.10526	2.3
SSE	0	0.03759	0	0	0	0	0.03759	2.3
S	0	0.03008	0	0	0	0	0.03008	2.3
SSW	0	0.03008	0	0	0	0	0.03008	2.3
SW	0.00752	0.03008	0	0	0	0	0.0376	1.99
WSW	0	0.00752	0	0	0	0	0.00752	2.3
W	0	0.00752	0	0	0	0	0.00752	2.3
WNW	0	0.01504	0	0	0	0	0.01504	2.3
NW	0	0	0	0	0	0	0	0
NNW	0	0.00752	0	0	0	0	0.00752	2.3
Totals	0.03008	0.96993	0	0	0	0	1.00001	2.25339

Stability Class B

Fraction of total meteorological data in stability class = 0.1255

	Windspeed (m s^-1^) 0.75	2.3	4.1	6.7	9.5	11	Fractional Direction Totals	Avg. windspeed (m s^-1^)
N	0.00478	0.01673	0.00876	0.0008	0	0	0.03107	2.68233
NNE	0.00637	0.04064	0.02789	0.00159	0	0	0.07649	2.91870
NE	0.01275	0.05976	0.0239	0	0	0	0.09641	2.54123
ENE	0.01833	0.06614	0.01594	0	0	0	0.10041	2.30279
E	0.02231	0.07809	0.02311	0.0008	0	0	0.12431	2.38476
ESE	0.02311	0.10279	0.03187	0	0	0	0.15777	2.43656
SE	0.01833	0.07012	0.04382	0.0008	0	0	0.13307	2.70568
SSE	0.01195	0.04064	0.02311	0.0008	0	0	0.0765	2.64765
S	0.01275	0.02709	0.01116	0.0008	0	0	0.0518	2.37423
SSW	0.01036	0.01514	0.00717	0	0	0	0.03267	2.20352
SW	0.00956	0.00717	0.00159	0.0008	0	0	0.01912	1.85878
WSW	0.00876	0.00717	0.00398	0	0	0	0.01991	1.97785
W	0.01195	0.00956	0.00637	0.0008	0	0	0.02868	2.17669
WNW	0.00717	0.00637	0.00239	0.0008	0	0	0.01673	2.10325
NW	0.00637	0.00478	0.00319	0.0008	0	0	0.01514	2.25961
NNW	0.00558	0.00717	0.00637	0.0008	0	0	0.01992	2.61812
Totals	0.19043	0.55936	0.24062	0.00959	0	0	1	2.48014

Stability Class C

Fraction of total meteorological data in stability class = 0.1734

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00173	0.01038	0.02653	0.00461	0	0	0.04325	3.81113
NNE	0.00404	0.0346	0.03979	0.00807	0	0	0.0865	3.46610
NE	0.00461	0.03172	0.0271	0.00346	0	0	0.06689	3.15002
ENE	0.00461	0.0271	0.01557	0.00173	0	0	0.04901	2.88136
E	0.00461	0.03806	0.0248	0.00404	0.00058	0	0.07209	3.12461
ESE	0.00519	0.04325	0.02941	0.00173	0	0	0.07958	2.95978
SE	0.00346	0.05133	0.05017	0.00634	0.00058	0.00058	0.11246	3.38537
SSE	0.00346	0.0496	0.05133	0.00807	0.00058	0.00058	0.11362	3.45966
S	0.00519	0.03806	0.03114	0.00519	0.00058	0	0.08016	3.23587
SSW	0.00461	0.02191	0.01384	0.00231	0	0	0.04267	2.95456
SW	0.00404	0.01615	0.01038	0.00288	0	0	0.03345	3.05019
WSW	0.00634	0.0075	0.01038	0.00519	0.00058	0.00058	0.03057	3.63840
W	0.00634	0.0173	0.01961	0.01038	0.00115	0.00058	0.05536	3.82581
WNW	0.00461	0.00865	0.01326	0.01038	0.00173	0.00173	0.04036	4.52749
NW	0.00288	0.01153	0.01845	0.00865	0.00058	0.00058	0.04267	4.08176
NNW	0.00288	0.0173	0.02595	0.00519	0	0	0.05132	3.56816
Totals	0.0686	0.42444	0.40771	0.08822	0.00636	0.00463	0.99996	3.40169

Stability Class D

Fraction of total meteorological data in stability class = 0.2752

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0	0.00291	0.0109	0.01999	0.00254	0.00218	0.03852	6.05986
NNE	0	0.00799	0.01054	0.01453	0.00109	0	0.03415	4.95745
NE	0	0.00654	0.00654	0.004	0.00073	0	0.01781	4.24430
ENE	0	0.00763	0.00436	0.00109	0	0	0.01308	3.26666
E	0	0.00836	0.00727	0.00581	0.00036	0	0.0218	4.19183
ESE	0	0.00654	0.00436	0.00254	0	0	0.01344	3.71547
SE	0	0.00908	0.00799	0.00509	0.00073	0	0.02289	4.13634
SSE	0	0.01235	0.0189	0.01344	0.00145	0.00036	0.0465	4.59522
S	0	0.01708	0.01853	0.02144	0.00218	0.00036	0.05959	4.75876
SSW	0	0.0149	0.00981	0.01199	0.00145	0.00036	0.03851	4.48088
SW	0	0.01308	0.01163	0.01708	0.00182	0.00036	0.04397	4.85451
WSW	0	0.01417	0.01381	0.03815	0.00836	0.00254	0.07703	5.87014
W	0	0.01635	0.01417	0.06323	0.03125	0.03379	0.15879	7.48100
WNW	0	0.01344	0.01417	0.11047	0.06214	0.0556	0.25582	7.93951
NW	0	0.01381	0.01453	0.05451	0.01817	0.00581	0.10683	6.48767
NNW	0	0.00872	0.01672	0.02398	0.00182	0	0.05124	5.20226
Totals	0	0.17295	0.18423	0.40734	0.13409	0.10136	0.99997	6.27112

387

Stability Class E

Fraction of total meteorological data in stability class = 0.1734

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0	0	0.02537	0.00173	0	0	0.0271	4.26597
NNE	0	0	0.02134	0.00058	0	0	0.02192	4.16879
NE	0	0	0.01211	0	0	0	0.01211	4.1
ENE	0	0	0.0075	0	0	0	0.0075	4.1
E	0	0	0.01096	0	0	0	0.01096	4.1
ESE	0	0	0.00807	0	0	0	0.00807	4.1
SE	0	0	0.01615	0	0	0	0.01615	4.1
SSE	0	0	0.03691	0.00115	0	0	0.03806	4.17856
S	0	0	0.08535	0.00173	0	0	0.08708	4.15165
SSW	0	0	0.08016	0.00115	0	0	0.08131	4.13677
SW	0	0	0.12111	0.00115	0	0	0.12226	4.12445
WSW	0	0	0.15398	0.00577	0	0	0.15975	4.19390
W	0	0	0.10438	0.00519	0	0	0.10957	4.22315
WNW	0	0	0.09343	0.00692	0	0	0.10035	4.27929
NW	0	0	0.10381	0.00288	0	0	0.10669	4.17018
NNW	0	0	0.08939	0.00173	0	0	0.09112	4.14936
Totals	0	0	0.97002	0.02998	0	0	1	4.17794

Stability Class F

Fraction of total meteorological data in stability class = 0.2381

	Windspeed (m s^{-1})						Fractional Direction Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00252	0.0168	0	0	0	0	0.01932	2.09782
NNE	0.00504	0.01638	0	0	0	0	0.02142	1.93529
NE	0.00546	0.02016	0	0	0	0	0.02562	1.96967
ENE	0.0084	0.01722	0	0	0	0	0.02562	1.79180
E	0.00882	0.02604	0	0	0	0	0.03486	1.90783
ESE	0.0084	0.02184	0	0	0	0	0.03024	1.86944
SE	0.00882	0.02898	0.00042	0	0	0	0.03822	1.96208
SSE	0.01176	0.04452	0	0.00042	0	0	0.0567	2.01111
S	0.0168	0.06174	0	0.00042	0	0	0.07896	1.99361
SSW	0.01554	0.06762	0.00042	0.00042	0	0	0.084	2.04425
SW	0.01596	0.08148	0.00042	0	0	0	0.09786	2.05493
WSW	0.01806	0.09618	0.00042	0.00084	0	0	0.1155	2.09618
W	0.0168	0.1134	0.00042	0.00084	0	0	0.13146	2.13578
WNW	0.01428	0.09282	0	0.00126	0	0	0.10836	2.14689
NW	0.01092	0.0714	0	0.00042	0	0	0.08274	2.11776
NNW	0.00798	0.04074	0	0.00042	0	0	0.04914	2.08589
Totals	0.17556	0.81732	0.0021	0.00504	0	0	1.00002	2.05388

388

Composite of all Stability classes								
	Windspeed (m s^{-1})						Direction Fractional Totals	Avg. windspeed (m s^{-1})
	0.75	2.3	4.1	6.7	9.5	11		
N	0.00149	0.00890	0.01309	0.00670	6.99008	5.99936	0.03149	4.23623
NNE	0.00269	0.01779	0.01700	0.00569	2.99968	0	0.04349	3.53334
NE	0.00369	0.02089	0.01159	0.00170	2.00896	0	0.03809	2.93183
ENE	0.00519	0.02159	0.00720	0.00059	0	0	0.03460	2.51795
E	0.00589	0.02730	0.01110	0.00239	1.99644	0	0.04690	2.78689
ESE	0.00580	0.02979	0.01169	0.00099	0	0	0.04829	2.64085
SE	0.00500	0.02849	0.01929	0.00260	3.01468	1.00572	0.05580	3.04324
SSE	0.00489	0.02819	0.02340	0.00549	4.99612	1.99644	0.06269	3.32161
S	0.00650	0.02980	0.02669	0.00730	7.00508	9.9072E	0.07109	3.36908
SSW	0.00579	0.02630	0.01999	0.00399	0.00039	9.9072E	0.05659	3.15415
SW	0.00580	0.02710	0.02630	0.00549	5.00864	9.9072E	0.06530	3.32627
WSW	0.00649	0.02910	0.03290	0.01259	0.00240	0.00079	0.08429	3.82823
W	0.00659	0.03579	0.02629	0.02040	0.00879	0.00939	0.10729	4.83505
WNW	0.00509	0.02829	0.02269	0.03380	0.01740	0.01560	0.12290	5.90207
NW	0.00389	0.02340	0.02559	0.01720	0.00510	0.00169	0.07689	4.47467
NNW	0.00309	0.01609	0.02540	0.00799	5.00864	0	0.05310	3.80132
Totals	0.07799	0.39889	0.32029	0.13499	0.03800	0.02869	0.99888	3.87038

385/389