Draft Task 3 Report: Calculation of Surface Radionuclide Soil Action Levels for Plutonium and Americium

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EPA
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DRAFT TASK 3 REPORT

I. Executive Summary

The Department of Energy (DOE), Environmental Protection Agency (EPA) and Colorado Department of Public Health and Environment (CDPHE) are calculating surface radionuclide soil action levels (RSALs) for plutonium and americium, which will guide soil remediation during the accelerated cleanup of Rocky Flats. These action levels will replace the levels established by the DOE, the EPA, and the CDPHE (the agencies) in 1996.

This report, Task 3, is the last of five reports that were prepared during this review and represents the culmination of the information developed in the other four reports. These other reports are Task 1: Regulatory Analysis; Task 2: Computer Model Selection; Task 4: New Science; and Task 5: Determining Cleanup Goals at Radiologically-Contaminated Sites.

The Task 3 report discusses the exposure scenarios that the agencies are using for the calculation of new RSALs, as well as the methods of calculation, the associated input parameters, and the results of the calculations. Dose calculations were performed using the RESRAD 6.0 (Residual Radioactivity) model and risk calculations were performed following EPA's standard risk methodology. Five exposure scenarios are addressed in this report: wildlife refuge worker, rural resident, open space user, office worker, and resident rancher. Plutonium and americium activity concentrations in surface soil were calculated for a 25 millirem (mrem) annual dose and for concentrations within EPA's target risk range of one in 10,000 to one in one million (10^{-4} to 10^{-6}) cancer incidence for various land use scenarios. The results are summarized in the table below:

Dose and Risk Calculations for Plutonium in Surface Soil Adjusted by Sum-of-Ratios Method[c] (pCi/g)

Land Use Scenario	Risk Levels			25-mrem annual dose
	10^{-4}	10^{-5}	10^{-6}	
Wildlife refuge worker [a]	490	49	5	862
Rural Resident – adult [a]	173	18	2	209
Rural Resident/sp– child [a]				244
Open Space User – adult [b]	1047	105	10	11797
Open Space User – child [b]				4842
Office Worker [b]	596	60	6	2289
Resident Rancher	-	-	-	45

[a] Percentile consistent with Reasonable Maximum Exposed individual

[b] Deterministic

[c] This example accounts for additional activity from Am-241 using a sum-of-ratios method, and assumes that the Am:Pu activity ratio equals 0.1527 and that only Am and Pu are present.

The dose estimate for the resident rancher scenario listed in the table above was calculated using the same parameters as those used by Risk Assessment Corporation (wherever possible) for the purpose of comparing the model software they employed to that used by the agencies.

When the technical peer review of this report is completed, the agencies will select RSALs based on the results of the analyses in this report. The analyses will also provide a basis for establishing final cleanup levels at Rocky Flats, taking into account other factors, such as the effort to clean up "as low as reasonably achievable" and impacts to long-term site stewardship.

II. Introduction for Draft Calculation for Surface RSALs for Plutonium and Americium

The agencies are proposing new Radionuclide Soil Action Levels (RSALs) for surface soil for plutonium and americium to guide the cleanup at Rocky Flats. These RSALs will replace those levels established in 1996. The RSALs are the activity concentrations of radionuclides such as plutonium and americium in soils, which, if exceeded, trigger an evaluation, a remedial action, or a management action. Existing RSALs are under review and new RSALs are being proposed for a number of reasons, including:

- The Rocky Flats Cleanup Agreement requires periodic review of action levels.
- The current RSALs have been controversial among local governments and community members.
- A draft radiation site cleanup rule that was used as the basis for the current RSALs was never promulgated.
- New technical information relevant to the RSALs has become available since the current RSALs were developed in 1996, including an independent calculation of RSALs by the Risk Assessment Corporation.

This report, Task 3, discusses the exposure scenarios that the agencies are using for the calculation of new RSALs, as well as the methods of calculation, the associated input parameters, and the results of the calculations. Five exposure scenarios are addressed in this report: wildlife refuge worker, rural resident, open space user, office worker, and resident rancher.

The agencies chose the wildlife refuge worker scenario because it appeared likely that Rocky Flats would be designated a national wildlife refuge. The rural resident scenario was chosen because the agencies believe that if institutional controls fail in the future, a residential scenario represents a foreseeable land use. Calculations based on the office worker and the open-space user were performed because the Rocky Flats Cleanup Agreement signed in 1996 listed those scenarios as anticipated future uses. These scenarios were evaluated primarily to provide a comparison to the 1996 RSALs. The agencies calculated a value for a resident rancher scenario using the same parameters as Risk Assessment Corporation (wherever possible) for the purpose of comparing the model software they employed to that used by the agencies and at the request of members of the public.

The primary regulatory bases for the Rocky Flats RSALs stem from the Nuclear Regulatory Commission decommissioning rule and the Superfund law (Comprehensive Environmental Response, Compensation and Liability Act) (for a more complete discussion of the regulatory bases, refer to the Task 1 report). The former says that the site should be cleaned up so that a future user will not receive a dose greater than 25 mrem/year and that residual radioactivity is reduced to a level "as low as reasonably achievable." Since the Nuclear Regulatory Commission rule is relevant to and appropriate for the cleanup of Rocky Flats, the agencies performed dose assessments to develop potential RSAL values that correspond to a dose of 25 mrem/year (millirem/year). RESRAD is the computer model used for that assessment. Earlier versions of RESRAD were used by the agencies in 1996 and later by the Risk Assessment Corporation. Since the 25 mrem/year dose limit may not meet the protective risk range spelled out in the

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Comprehensive Environmental Response, Compensation and Liability Act of one in ten thousand to one in a million (10^{-4} to 10^{-6}), the agencies also developed potential RSAL values based on risk using the EPA's standard risk equations.

Principle changes in methodology between the 1996 calculations and the current effort are the use of probabilistic methodologies in the calculations in contrast to the purely deterministic methods employed in 1996 and the use of updated dose conversion factors and cancer slope factors. Differences between a deterministic analysis and a probabilistic analysis can be summarized as follows:

- Deterministic (point estimate): Single parameter values are used in an equation to calculate a value, in this case a concentration of radionuclides in the soil that equates to a target dose level or risk level (e.g. 25 mrem/year or one in 10,000);

- Probabilistic: For highly sensitive parameters, distributions of values are substituted for single point values and the equation is solved over and over with computer software that randomly chooses different values from the input distributions for each iteration. Hundreds or thousands of iterations are performed to produce an output that is itself a distribution. In this case that output distribution represents various levels of contamination that could result in a target dose or risk level depending on the variability of important exposure parameters such as inhalation rate and time spent on site. EPA guidance specifies that the RSAL should be a value corresponding with the reasonably maximally exposed individual of that output distribution.

The agencies spent considerable effort in determining the sensitive parameters, evaluating whether parameters should be represented by deterministic values or probabilistic distributions for those parameters, and entering those values into the selected dose and risk modeling equations. This report provides the results of RSAL calculations for the five scenarios discussed above. For the office worker, open-space user, refuge worker, and rural resident scenarios, results are provided in picocuries/gram of soil that equate to the target dose of 25 mrem/year and the risk levels of one in 10,000 (10^{-4}), one in 100,000 (10^{-5}), and one in 1,000,000 (10^{-6}).

Section III provides detailed discussions of the four land use scenarios employed for both dose and risk assessments: wildlife refuge worker, rural resident, open space user and office worker.

Section IV gives the reader overview information on dose and risk analysis, discusses the method of conducting a pathway and parameter sensitivity analysis, and presents the results of those analyses. It also discusses the process for developing parameter distributions, provides detail on the derivation of the mass loading distribution, and gives the rationale for the selection of cancer slope factors and the dose conversion factors.

Section V presents the results from the dose and risk assessments for the wildlife refuge worker, rural resident, open space user, and office worker scenarios.

Section VI provides a discussion of the variability and uncertainty of the dose and risk assessments, as well as a qualitative discussion of the level of conservatism.

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The following appendices supply information about the methods of calculation and the parameters used:

- Appendix A: Justification and Supporting Documentation for Input Parameters
- Appendix B: Description of the Standard Risk Equations
- Appendix C: Risk Based Spreadsheets and Instructions For Use
- Appendix D: Complete RESRAD Input Parameters for Dose Calculations
- Appendix E: RESRAD Modeling Outputs (Available on CD Rom upon Request)
- Appendix F: PM10 Air Monitoring Data from Rocky Flats and the State of Colorado
- Appendix G: RESRAD Results for the Resident Rancher Scenario

III. Scenario Selection for Dose and Risk Assessments

This section describes each of the land use scenarios that were evaluated for this study. A comparison of the features of each of the scenarios is summarized in Table III-2, Scenario Features Comparison Chart. Physiological and site specific physical parameters common to all scenarios are described separately in Section V. For all pathways described in these scenarios a sensitivity analysis has been conducted on each of these pathways as well as on the combination of all potentially active pathways to identify those input parameters that influence the output doses.

Figures III-1 through III-4 provide conceptual site models that delineate the potential pathways for exposure to contaminants for each exposure scenario. The conceptual site models identify which of the exposure pathways are considered complete, i.e., capable of transferring harmful effects from radionuclides in surface soils to exposed individuals. The complete pathways are further identified as either significant or insignificant, based on their contribution to the calculated dose or risk. Table III-1 compares these pathways for the exposure scenarios.

TABLE III-1 Summary of Complete Pathways for Each Exposure Scenario

	EXPOSURE SCENARIOS			
PATHWAYS	**Wildlife Refuge Worker**	**Rural Resident**	**Open Space User**	**Office Worker**
Surface water ingestion	I	I	I	IC
Surface water-dermal contact	I	I	I	IC
Soil/sediment ingestion	S	S	S	S/IC
Soil/sediment-dermal contact	I	I	I	I/IC
Plant ingestion	IC	S	IC	IC
Dust inhalation	S	S	S	S
External gamma irradiation	S	S	S	S

S = significant pathway
I = insignificant pathway
IC = incomplete pathway

The agencies chose the wildlife refuge worker scenario because it appeared likely that Rocky Flats would be designated a national wildlife refuge. Should institutional controls fail in the future, a residential scenario is a foreseeable land use. Calculations based on the office worker and the open-space user were performed because the Rocky Flats Cleanup Agreement, signed in 1996, listed those scenarios as anticipated futures uses. These scenarios were evaluated primarily to provide a comparison to the 1996 RSALs. The agencies calculated a value for a resident rancher scenario (see Appendix G) using the same parameters as Risk Assessment Corporation, wherever possible, for the purpose of comparing the model software used by Risk Assessment Corporation to that used by the agencies, and at the request of members of the public.

III-1) Scenario Descriptions

a) Wildlife Refuge Worker

This scenario assumes that a wildlife refuge will be established on the acreage that is now Rocky Flats as a result of legislation that has been introduced into Congress. In accordance with the proposed existing legislation and guidance for a Rocky Flats Wildlife Refuge, the purposes of the proposed refuge are: 1) restoring and preserving native ecosystems; 2) providing habitat for, and providing management of, native plants and migratory and resident wildlife; 3) conserving threatened and endangered species; 4) providing opportunities for compatible, wildlife-dependent environmental scientific research; and 5) providing the public with opportunities for compatible outdoor recreational and educational activities. Given the proposed legislation for a wildlife refuge at Rocky Flats and the widespread community preference for preservation of Rocky Flats as open space, the wildlife refuge worker scenario represents the most likely future use of Rocky Flats.

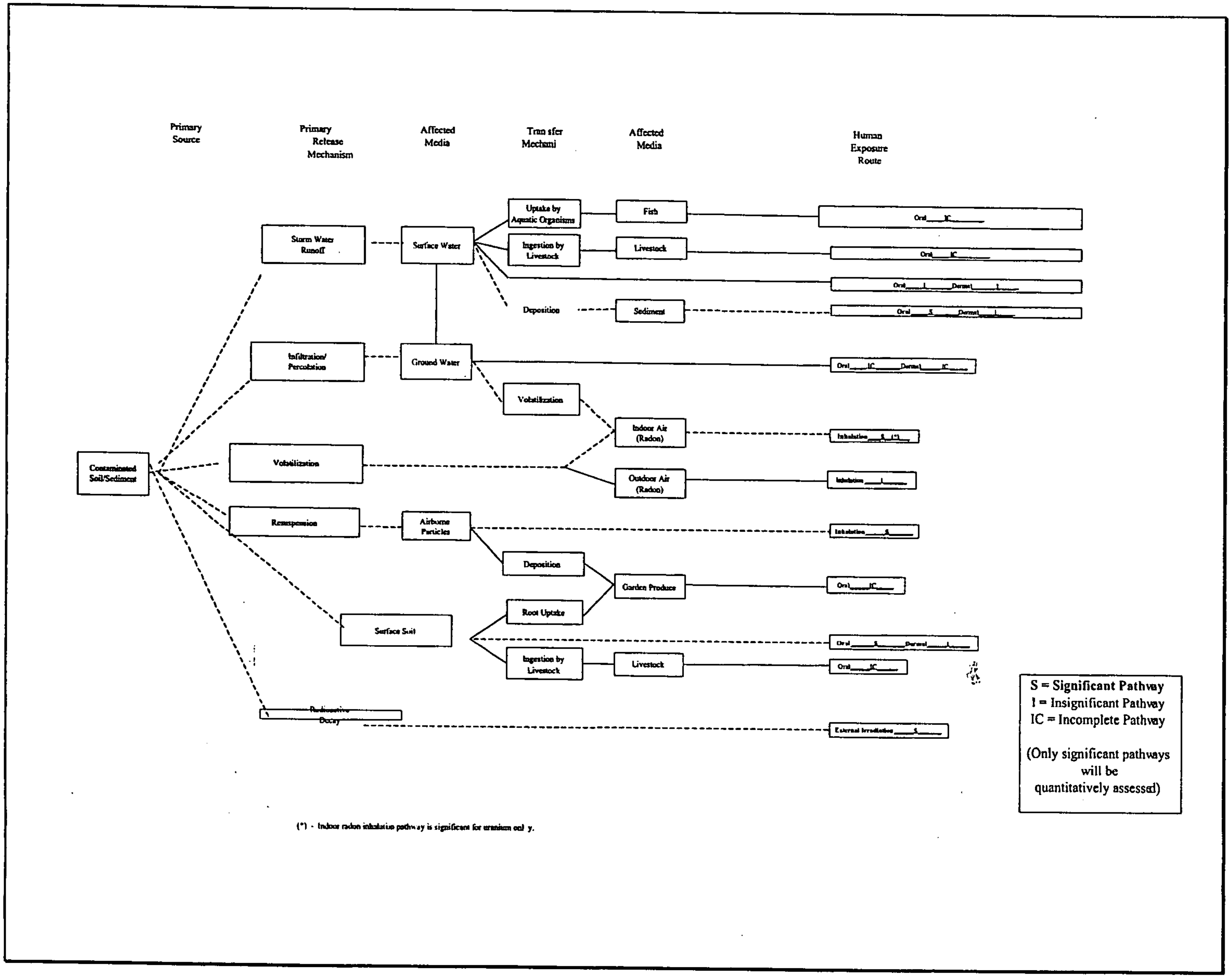
The scenario assumes that the refuge headquarters, which could include office buildings and equipment storage and maintenance shops, would be placed in that portion of Rocky Flats where soils contain residual contamination. It is assumed no visitor center would be developed at Rocky Flats, and facilities for childcare are not included as a part of the refuge building complex.

This scenario assumes that the wildlife refuge workers may be scientists, maintenance workers, equipment operators, or other occupations that require the worker to spend 100 percent of work time on-site and 50 percent outdoors. The wildlife refuge workers would spend all of their time on the contaminated area. The area is considered to be undeveloped surface soil with only vegetative cover over the contaminated soils except where buildings are present. Cover from lawn grasses, which would reduce exposure, has not been used in this or any other scenario. Refuge workers would perform a variety of activities where they could be directly exposed to surface or subsurface soil, breathe contaminated dust, and be exposed to external gamma radiation. Some of the tasks they do would involve physical labor resulting in an increased breathing rate and soil disturbing activities, which result in increased dust inhalation and increased soil ingestion. Windblown contaminated soil particles may be significantly increased during some days due to grass fires that have occurred on contaminated parts of the refuge. Refuge workers are assumed to work eight hours per day for five days per week and for 50 weeks each year.

It is assumed that the windows and doors of the buildings would be closed during cooler seasons, providing partial shielding from dust. During time indoors, the refuge worker would be partially shielded by the building from gamma radiation. There is no onsite source of fruits, vegetables, or drinking water that would be consumed by refuge workers.

The conceptual site model in Figure III-1 evaluates all of the possible pathways for contamination to reach this receptor and illustrates which pathways provide access to the receptor.

Figure III-1 Conceptual Site Model for Wildlife Refuge Worker



Exposure Pathways

Exposure pathways are conduits through which contaminants might travel from the environment to a receptor, in this case the wildlife refuge worker. Active pathways are those that are realistic and which contribute to the dose and risk in model outputs. There are three active exposure pathways that are considered complete and potentially significant for the wildlife refuge worker scenario: ingestion of contaminated soil, inhalation of contaminated dust, and external exposure to gamma radiation from contaminated surface soil. The three active pathways were determined by applying the site conceptual model to remove any non-applicable pathways. These three exposure pathways were quantitatively assessed in deriving an RSAL for the wildlife refuge worker.

Pathways that would not be complete or significant for the worker have been excluded. For instance, the consumption of contaminated garden fruits and vegetables and the consumption of contaminated shallow groundwater as drinking water have been excluded for the wildlife refuge worker scenario because these pathways are not viable. While it could be argued that a worker could discover wild fruits or ingest surface water on the refuge, such incidents would be rare and would not be a significant contributor to a realistic exposure scenario. Pathways requiring consumption of meat, milk or aquatic food produced on the refuge (none realistically available), or those requiring exposure to radon, tritium and carbon 14 (attributable only to natural background) have also been excluded.

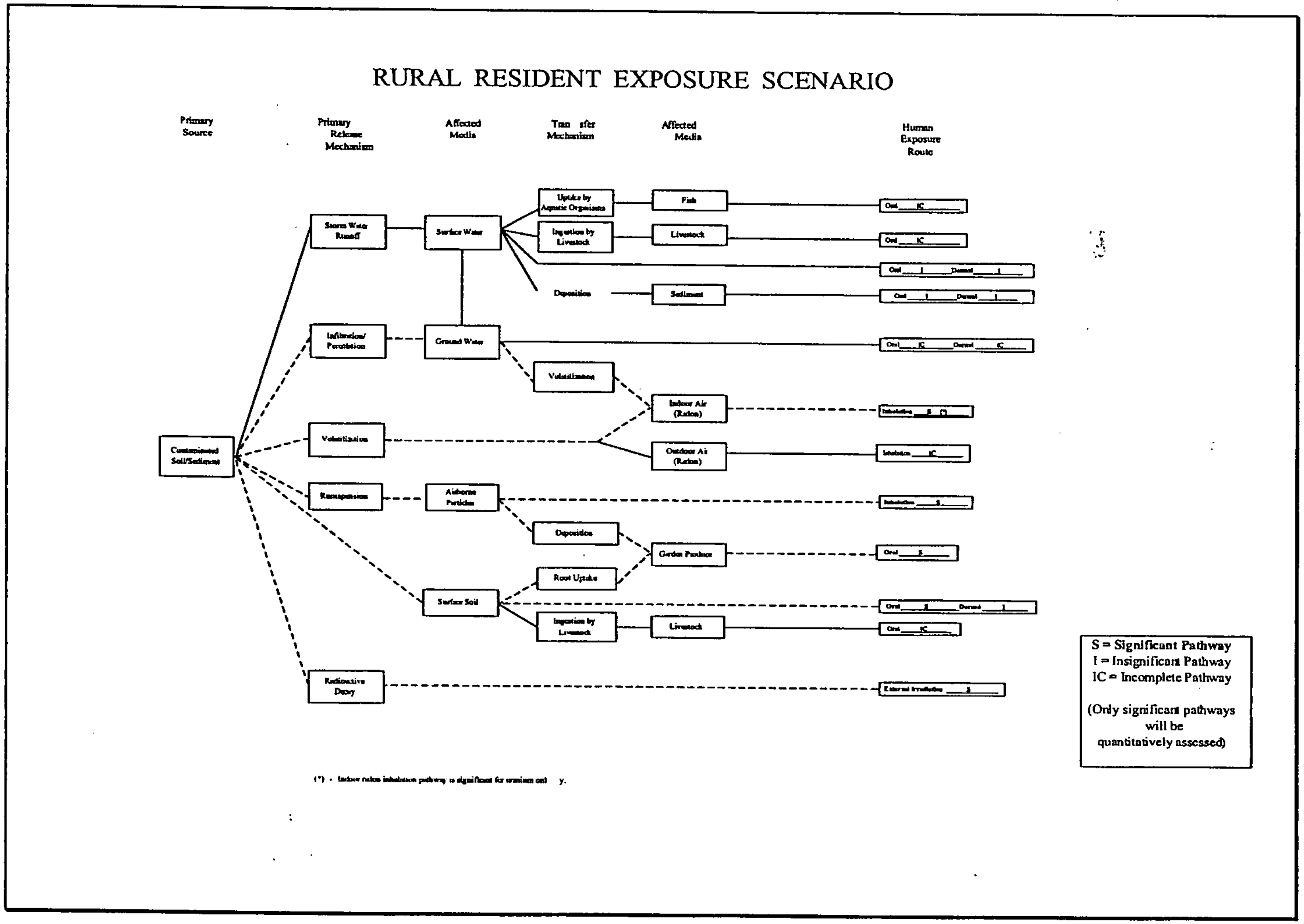
b) Rural Residential Scenario

A rural residential scenario was chosen to represent a future user of the Rocky Flats Industrial Area in the event that institutional controls fail or are not present to prevent the occupation of areas with contaminated soils. Residents considered in this scenario are adults and children who would spend most of their time on-site and up to 20 percent of their time outdoors. The indoor exposure rate from gamma radiation would be reduced by the building structures, and the contaminated dust present in outdoor air would be present in indoor air at a reduced concentration commensurate with having windows closed during cool weather. Dust occasionally would be increased by fires that burn off the accumulated vegetation.

The entire residential site and large surrounding areas are assumed to be uniformly contaminated with plutonium and americium at the RSAL concentration values. Residents are assumed to spend up to 350 days per year on-site for 24 hours per day for up to 40 years. The residents would live on 5 acre sites with undeveloped surface soils and native vegetative cover over contaminated soils. Cover from lawn grasses, which would reduce exposure, have not been used in this or any other scenario. Homegrown produce would be ingested, but no shallow groundwater would be consumed as drinking water.

Figure III-2 provides a conceptual site model that delineates the potential pathways for exposure to contaminants by a resident.

Figure III-2 Conceptual Site Model for Rural Resident



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Exposure Pathways
The active pathways associated with the rural resident scenario are ingestion of surface soil/indoor dust; ingestion of contaminated homegrown produce; inhalation of surface soil/indoor dust particles; and external exposure to gamma radiation. The active pathways are those pathways which are deemed realistic and which contribute to the dose in the RESRAD model outputs. The four pathways listed above were determined by performing a pathway sensitivity analysis for a residential user and then applying the site conceptual model to remove any non-applicable pathways.

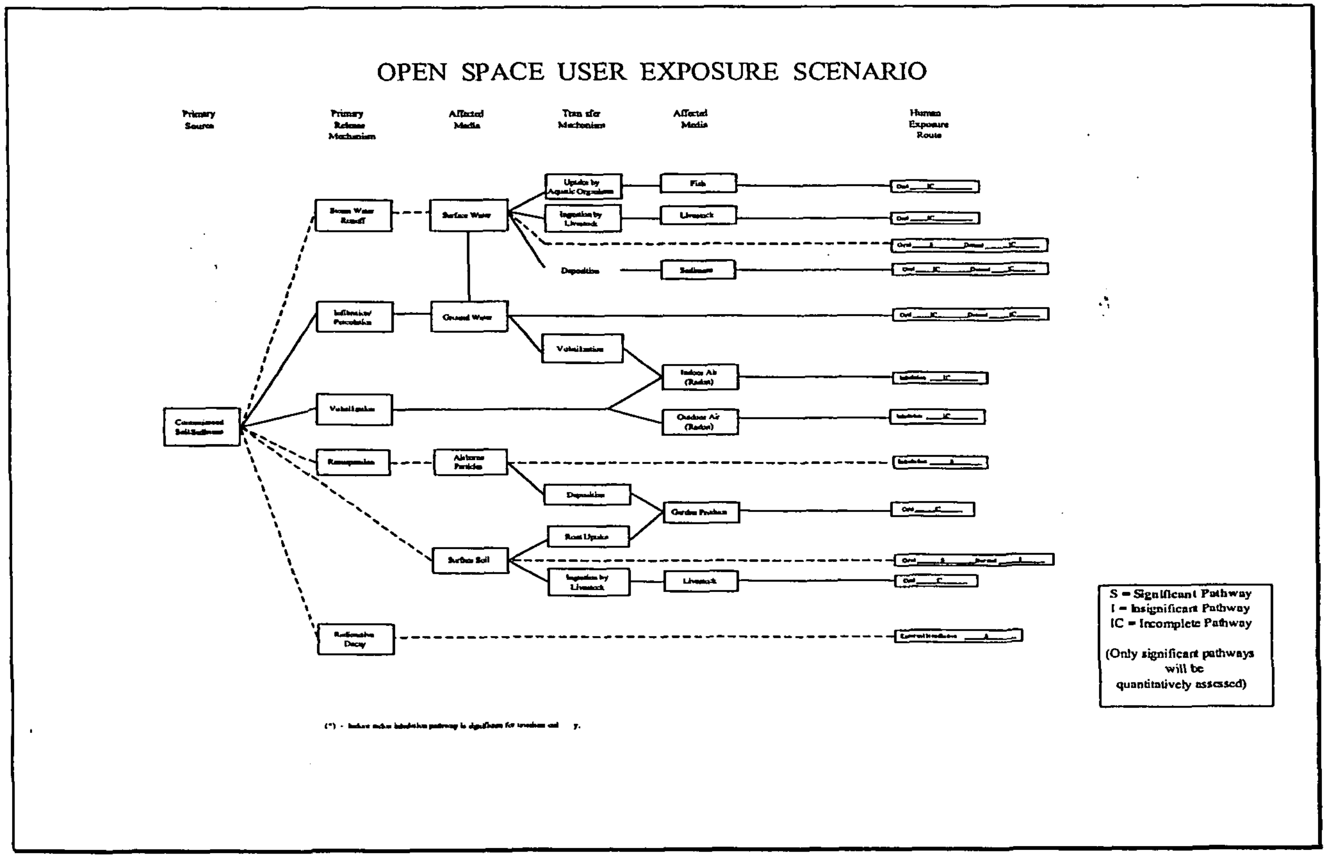
Pathways that would not be complete or significant for the resident have been excluded. The pathways of consumption of shallow groundwater, consumption of meat, milk and aquatic food from the site, and exposure to radon, tritium and carbon 14 (attributable to natural background only) were suppressed because they are not believed to be viable contributors for this scenario.

c) Open Space User Scenario
The open space user scenario represents a future user of Rocky Flats who visits the site for occasional recreation. This scenario is one of several potential uses identified in RFCA after cleanup is completed. This scenario describes a site, which remain as open space and would not be developed in the future. The open space scenario anticipates access by the public to the Buffer Zone in a manner similar to other open spaces currently used nearby in Jefferson and Boulder counties. For example, the time an open-space user spends on site in this scenario is consistent with recent survey data from the counties (Jefferson County, 1996; Boulder County, 1996).

Open space users, both children and adults, would visit the open space 100 times per year and spend 2.5 hours per visit, all outdoors. It is assumed that local residents could visit the site over a period of 30 years. No fruits, vegetables, or water originating from on-site would be routinely ingested. Native vegetative cover would be present over the entire open space area, except in the aftermath of a prairie fire. Concentrations of windblown contaminated soil particles increase significantly during some visits due to fires that would have occurred on contaminated parts of the open space. All visits are assumed to be confined to a uniformly contaminated area at RSAL concentrations.

Figure III-3 provides a conceptual site model that delineates the potential pathways for exposure to contaminants by a visiting open space user.

Figure III-3 Conceptual Site Model for Open Space User



Exposure Pathways

There are three exposure pathways that are considered complete and potentially significant for the open space user scenario: soil ingestion, dust inhalation, and external gamma exposure from contaminated surface soil. These exposure pathways will be quantitatively assessed in deriving an RSAL for the open space user.

Pathways that would not be accessible to the user have been excluded. For instance, the consumption of contaminated garden fruits and vegetables and the consumption of contaminated shallow groundwater as drinking water have been excluded for the open space user scenario because these pathways are not viable. Pathways requiring consumption of meat, milk or aquatic food grown on site, or those requiring exposure to radon, tritium and carbon 14 (attributable only to natural background) have also been excluded.

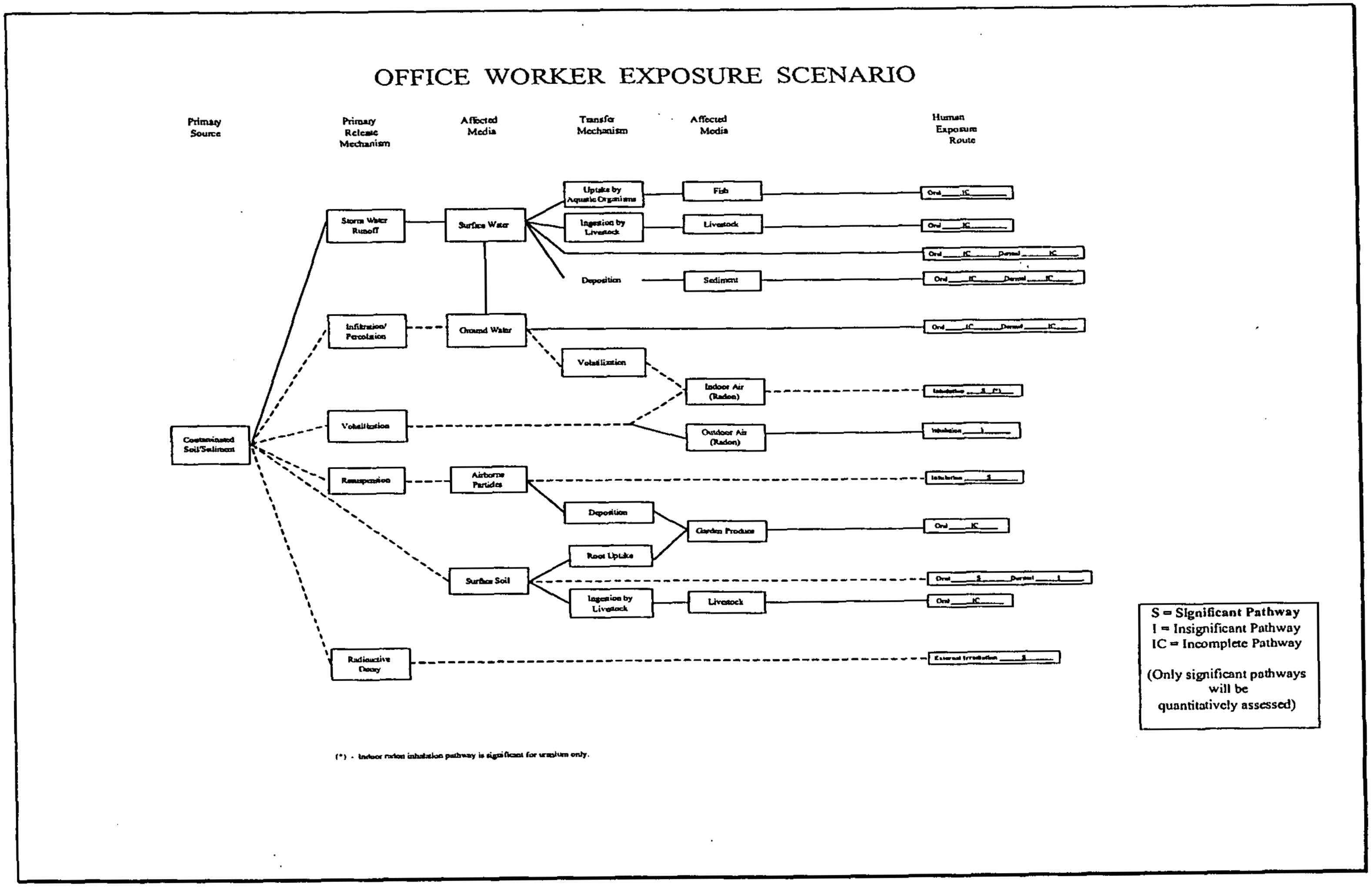
d) Office Worker Scenario

An office worker scenario was chosen to represent a potential future user after cleanup because RFCA lists commercial/industrial development as a possible future use for Rocky Flats. Office workers considered in this scenario are adult men and women working in an administrative environment, spending 100 percent of their time indoors. Time on-site would be eight hours per day, five days per week for 250 days or 2000 hours per year. Workers are assumed to spend 25 years working at the site.

The commercial/industrial development area where the offices would be located is the contaminated area, most of which is undeveloped surface soils with only native vegetative cover over contaminated soils. Office workers would be exposed to soil indirectly via ingestion and inhalation of indoor dust assumed to infiltrate through the building's ventilation system. Dust in the air would be increased occasionally by grass fires that burn off the vegetation. The office workers would be partially shielded from gamma radiation from surface soils due to building structures. Office workers would not consume fruits, vegetables, or shallow groundwater that originate at the site.

Figure III-4 provides a conceptual site model that delineates the various potential pathways for exposure to contaminants by an office worker.

Figure III-4 Conceptual Site Model for Office Worker



<u>Exposure Pathways</u>

The active pathways associated with this scenario are incidental ingestion of surface soil/indoor dust; inhalation of surface soil/indoor dust particles; and external exposure to gamma radiation. The active pathways are those pathways which are deemed realistic and which contribute to the doses in the RESRAD model outputs. A sensitivity analysis has been conducted on each of these pathways as well as on the combination of all three pathways to identify those RESRAD input parameters that influence the output doses.

The consumption of contaminated garden fruits and vegetables and the consumption of contaminated shallow groundwater as drinking water were excluded for the Office Worker scenario because these pathways do not exist for an office worker. In addition, the pathways requiring consumption of meat and milk and aquatic food grown on site, and exposure to radon, tritium and carbon 14 (attributable to natural background only) were suppressed because they are not applicable to the scenario.

Table III-2 Scenario Features Comparison Chart

This table compares the physical conditions that make up each scenario and affect the exposure that users would receive. While there are differences between all of the scenarios, there are also conditions that the scenarios have in common.

Scenario Features	**Refuge worker**	**Office Worker**	**Open Space**	**Rural Resident**
Radiation dose limit	25 mrem/yr	25 mrem/yr	25 mrem/yr	25 mrem/yr
Risk level	calculated at 1 E^{-4}, 1 E^{-5} and 1 E^{-6} target levels	calculated at 1 E^{-4}, 1 E^{-5} and 1 E^{-6} target levels	calculated at 1 E^{-4}, 1 E^{-5} and 1 E^{-6} target levels	calculated at 1 E^{-4}, 1 E^{-5} and 1 E^{-6} target levels
Time on-site	variable up to 250 days/yr, 8 hours per day; 5 days per week	250 d/yr, 8 hours per day; 5 days a week	30 times per year and 3 hours per visit	variable up to 350 days/yr at 24 hours per day
Percent of on-site time outdoors	50%	0%	100%	20%
Life time at the site	7 to 14 years	25 years	30 years	up to 40 years
Cover over contaminated soils	native vegetation	native vegetation	native vegetation	native vegetation
User activity level	sedentary and active	sedentary	active	sedentary and active
On-site fruits or vegetables	none	none	none	yes
onsite drinking water source	none	none	none	none
Windows and doors	open during warm weather	closed with ventilation	no indoor exposure	open during warm weather
Indoor exposure rate from gamma radiation	40% of outdoor rate	40% of outdoor rate	none	40% of outdoor rate
Increased airborne contamination after fires	yes	yes	yes	yes

Note: See Appendix D for the detailed descriptions of the values used in RESRAD.
See Appendix C for the risk-based spreadsheet.
See Appendix A for a detailed description of the probabilistic distributions.

III-2) Exposure Pathways with Insignificant Contributions to Dose or Risk

A number of potential pathway analyses have been excluded for this RSAL analysis. These pathways are excluded either because the pathway is not linked physically between the source and the potential receptor, or because the potential dose from the pathway is insignificant compared to the primary pathways. This section describes the rationale for excluding certain pathways as contributors to dose or risk for future exposed individuals at Rocky Flats.

Direct Dermal Absorption Contact Pathway
In risk analysis, transfer of contaminants to a receptor through contact with the skin is a potential pathway associated with surface soil, sediments, or contaminated water. Dermal contact is considered to be a complete but insignificant pathway. Although some receptors will have direct contact with the soil and water, plutonium and americium will not be absorbed through intact skin. In all scenarios, drinking water and irrigation water, if used, would be provided from reliable deep wells or from commercial water systems. Direct contact with surface water would be only incidental in any of the scenarios.

Inhalation of Gases
The presence of gaseous radionuclides provides a potential contaminant exposure route to humans. Neither plutonium nor americium contribute gaseous daughter products that can lead to contaminant exposure. This pathway will be considered in later discussions of uranium daughter products, specifically radon. This exposure pathway will not be assessed for plutonium or americium isotopes.

Ingestion of Surface Water, Ground Water, and Food
Candidate exposure routes to humans from surface-water related contaminant sources include the potential ingestion of surface water. Ingestion of surface water is considered a complete pathway since individuals who visit or inhabit the site could splash water into their mouths or drink the raw water during a visit or sojourn across Rocky Flats. The availability of water is limited and the incidence of raw surface water ingestion by any of the users defined in these scenarios would be rare, resulting in an insignificant pathway. Surface water is ephemeral in the streams affected by surface contamination and cannot be considered a reliable source of water for drinking or other domestic purposes.

Potential contaminant exposure routes for groundwater include oral ingestion of lower hydrostratigraphic unit groundwater or upper hydrostratigraphic unit groundwater. Groundwater contribution to dose and risk is considered part of an incomplete, or at worst, insignificant, pathway. The only exposed individual who would potentially use shallow groundwater as a drinking source would be the rural resident. This scenario, does not assume a subsistence existence, but instead the rural resident lives on a five-acre plot and uses potable water derived either from a deep well or from a domestic water system.

A recent white paper (RMRS, 2001) concluded that it might be possible for wells at Rocky Flats to provide sufficient water for subsistence quantities of drinking water. However, the study was limited to looking only at the potential yields of wells that were unaffected by any other withdrawal of water from that same shallow source, and included imported water now leaking into and potentially contributing to the shallow water table. The working group concluded that such wells could not provide enough water for domestic use on a sustained basis. The potentially contaminated shallow groundwater supply would not be sufficiently reliable to be used routinely nor would such use be acceptable practice. In none of the scenarios defined would the exposed individuals be expected to have access to or use groundwater. These pathways were not quantitatively assessed in those four scenarios.

The ingestion of contaminated food products, other than fruits and vegetables, is an incomplete pathway in all scenarios and will not be quantitatively assessed. Fish living on site in the ephemeral streams are too small to be fished or eaten. Livestock grazing would not be viable on the small plots allocated for the rural resident, except when fed large quantities of purchased grains and hay grown elsewhere. The uptake of contaminants by livestock through limited incidental grazing is not likely to be a significant contributor to potential dose. These pathways are considered incomplete and will not be quantitatively assessed.

III-3) Solubility of Plutonium and Americium

Plutonium and Americium in Water

The mobility of environmental plutonium and americium in water is severely limited due to the extremely low solubility of these materials. At Rocky Flats, the plutonium is commonly identified as weapons grade plutonium. Americium in this environment is associated with that same material, as a result of ingrowth (decay) from plutonium-241 to americium-241. The RESRAD groundwater transport calculations treat plutonium and americium separately, and do not adequately represent the behavior of weapons-grade material containing both. RESRAD will overestimate the contribution of americium in this environment. In an ambient environment, plutonium rapidly forms an oxide; the small quantity of americium associated with that plutonium will generally be contained within the same particulate matrix.

Actinide migration studies at Rocky Flats have shown that the plutonium found in surface water is transported not as dissolved molecules but as particles of plutonium oxide, attached to colloids of organic material smaller than a 0.45 micron pore size filter. Typically, elevated concentrations of plutonium that have been observed in surface water runoff are not observed downstream of the detention ponds at the Site. The detention ponds are very effective in reducing the concentration of plutonium, due to settling of the particulate material in the pond sediments.

Plutonium has only been found in a few shallow groundwater wells at Rocky Flats. When found, it does not appear in nearby wells in patterns that appear to be attributable to plumes. This observation suggests that the plutonium is more likely due to contamination introduced into the well from surface contamination carried down during the construction of the well, but to date that hypothesis has not been confirmed. Tests continue to increase understanding of the presence of plutonium and americium in groundwater wells. Plutonium contamination in groundwater appears to be possible but not predictable. Although uncertainty exists concerning the potential of plutonium to move to ground water, the working group has concluded that the limited availability of shallow groundwater diminishes the impacts on future residents.

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IV. Selection of Input Parameters for Dose and Risk Calculations

Potential RSALs were calculated based on both dose, the energy from ionizing radiation received by target organs in the body, and risk, the likelihood of getting cancer. The dose-based calculations were performed using the equations and parameters in the RESRAD computer model, and the risk-based calculations were performed using the EPA's standard risk assessment methodology (U.S. EPA, 1989, U.S. EPA 1991, U.S. EPA 2001a). Both dose and risk methods use mathematical formulas to estimate the amount of toxic substance that a hypothetical individual is exposed to. The dose assessment method then multiplies the amount of exposure by a dose conversion factor to arrive at a predicted dose. The risk assessment method multiplies the amount of exposure by a cancer slope factor to arrive at a probability of risk. Appendix B describes the equations and parameters used in the risk-based approach for each land use scenario (e.g., residential, wildlife refuge worker) and for each exposure pathway (e.g., soil ingestion, inhalation). Appendix D describes the RESRAD model and parameters. An example of a risk-based RSAL equation for soil ingestion is shown below:

$$RSAL = \frac{TCR(e.g. 10^{-6})}{SISF \times SIR \times EF \times ED \times 0.001}$$

Where:

TCR = Target Cancer Risk
SISF = Soil Ingestion Slope Factor
SIR = Soil Ingestion Rate
EF = Exposure Frequency
ED = Exposure Duration

The equation consists of parameters for exposure variables, such as intake rates and exposure frequency, and toxicity variables such as the cancer slope factor. These parameters can be described by either single values or by a range or distribution of values. For example, the number of years an individual may reside on a contaminated site can be described as 30 years or as a range from one to 40 years.

If the potential RSAL is calculated using only single values or point estimates to represent each parameter, this approach is referred to as a point estimate approach (also called deterministic approach). The output or RSAL value from this approach will be a single value. If one or more of the parameters in the equation are represented by a distribution of values, otherwise known as probability distributions, this is referred to as a probabilistic approach. When one or more of the equation inputs are probability distributions, the output will be a distribution of soil action levels. If the input distributions represent variability in the magnitude and duration of exposure, then the output distribution can provide information on variability in risk in the population of concern. Figure IV-1 below illustrates the input of probability distributions into a soil action level equation and the resulting distribution of soil action levels.

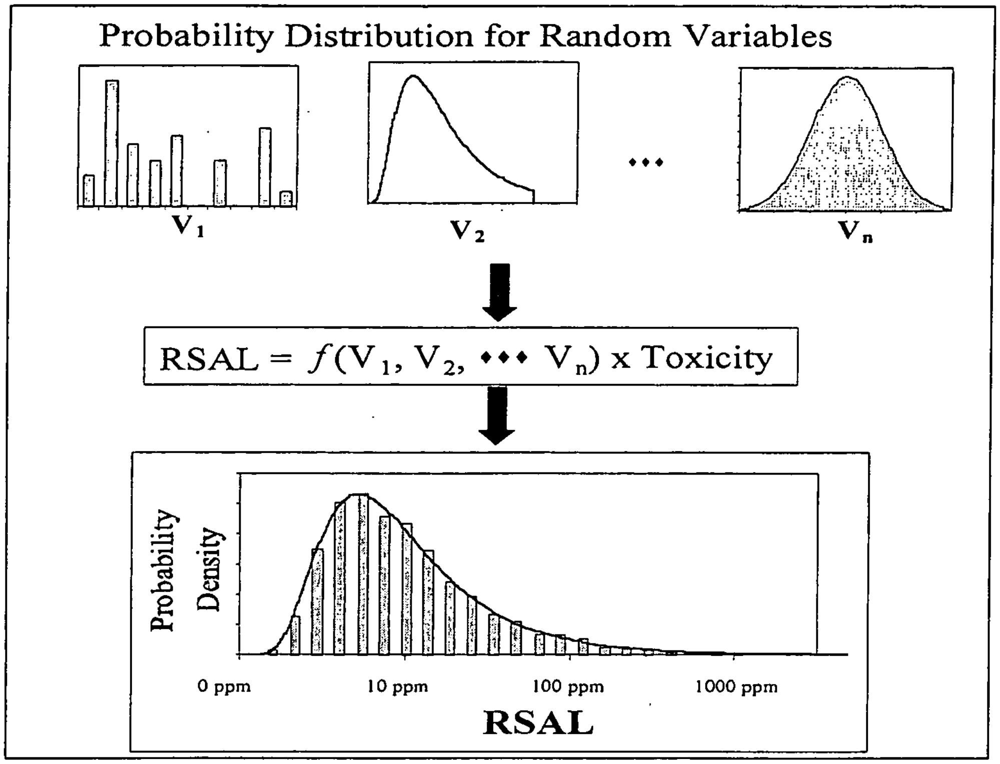

Figure IV-1. Conceptual model Monte Carlo analysis. Random variables (V_1, V_2,...V_n) refer to exposure variables (e.g., body weight, exposure frequency, ingestion rate) that are characterized by probability distributions. A unique radionuclide soil action level (RSAL) estimate is calculated for each set of random values. Repeatedly sampling $\{V_i\}$ results in a frequency distribution of soil action levels, which can be described by a probability distribution.

In the RSAL calculations, the exposure parameters for each pathway were assessed in terms of their impact on the outcome (the RSAL result) and a decision was made to either use probability distributions or point estimates. EPA policy recommends against developing site-specific probability distributions for human health toxicity values at this time, so point estimates were used for dose conversion factors and cancer slope factors (U.S. EPA, 2001b). These toxicity values are discussed in detail in Sections IV-7 and IV-8.

IV-1) Description of the Process for Selection of Initial Parameter Inputs

After looking at the conceptual site models in Section III of this report, it is immediately apparent that there are a large number of scenarios, pathways of exposure, and exposure and fate/transport variables that must be evaluated at the Rocky Flats site. Selecting and fitting probability distributions for all of these variables can be time and resource intensive, and is generally unnecessary. Therefore, it is important to identify factors that have a strong influence on the outcome early in the process. The use of sensitivity analysis is invaluable in identifying which variables and pathways most strongly influence the RSAL estimate.

Section IV-2 describes in detail the process used to conduct the sensitivity analysis. The process was used first to elucidate the most influential pathways of exposure and then to find the most influential variables within each pathway. The results of those sensitivity analyses are shown in

Section IV-3. For those variables identified as influential, the RSAL workgroup evaluated the existing data to determine if a probability distribution could be developed. If the data were deemed adequate, a distribution was developed. If they were not, a health protective point estimate was selected. The inputs selected for each of the influential variables are described in detail in Appendix A. It is important to note that when a sensitivity analysis is performed and the major variables elucidated, this does not mean that the less influential pathways and variables are eliminated from a risk assessment. They are kept in the assessment, typically as point estimates. For those variables, which were not identified as being influential, the default point estimates in RESRAD 6.0 and the most recent point estimates in the 1996 Preliminary Remediation Goals spreadsheets were used. These are described and shown in Appendix C. These combinations of probability distributions and point estimates were used to calculate the probabilistic RSALs.

Sensitivity Analysis

As the working group began developing input parameters for use in the RESRAD model and the Standard Risk Assessment Methodologies calculations, the group systematically tested RESRAD's response to changes in the various input parameters. Such a test, referred to as a sensitivity analysis, is generally used to identify the suite of input parameters that cause the greatest response in the model's output, in this case, the predicted dose or risk. This analysis shows the modeler those parameters whose influences are most important to the modeled results. The analysis allowed the working group to better focus its resources toward more accurate characterization, discussion, and validation of these more important parameters.

This resource issue was quite important to the working group. The effort to understand the origin, quality, and representativeness of the data that are used to determine a parameter input can be quite intensive. Limiting the number of parameters that must undergo this level of scrutiny was very important to the group due to time constraints.

Sensitivity Analysis Process

This section describes the sensitivity analysis process used in RESRAD 6.0; Standard Risk Assessment Methodologies parameters were scrutinized in the same manner using Crystal Ball. RESRAD 6.0 provides a Sensitivity Analysis module to assist the user who wants to perform such an analysis. The sensitivity analysis is centered around an initial input value for each parameter. The initial input parameters, or baseline values, were selected from values used in the 1996 RSAL analysis, except in cases where new information or new model requirements drove changes. Baseline values were reviewed prior to performing the analysis to ensure the baseline value and the resulting range of variability on that value were physically plausible and were compatible with the computational capabilities of the models. Using the module, input parameters can be varied to provide inputs ranging from some fixed fraction of the baseline parameter value to an equal multiple of the same baseline. For example, a parameter can be varied from one-third baseline to three times baseline, or from one-tenth to ten times, etc. For these extremes, the model is exercised keeping all other parameters constant, and the resultant doses are recorded. The relative change in dose can then be compared to the relative change in input value, and the effect of the change interpreted.

The working group performed the RESRAD sensitivity analysis separately for each pathway that would be active in the rural resident scenario, varying each active parameter in the pathway. The analysis was also conducted on the combination of all active pathways, so that the net influence of parameter variation across all pathways could be assessed. The rural resident scenario was used for this analysis since it contains the most comprehensive set of active pathways, and is the one that is likely to provide the most limiting contaminant results. Active pathways for the rural resident sensitivity analysis scenario are listed in Table IV-1.

Table IV-1 Sensitivity Analysis -- Pathways

RESRAD 6.0 Pathways	Pathway Active or Suppressed
External Gamma	Active
Inhalation	Active
Plant Ingestion	Active
Meat Ingestion	Suppressed
Milk Ingestion	Suppressed
Aquatic Foods	Suppressed
Drinking Water	Suppressed
Soil Ingestion	Active
Radon	suppressed

The working group originally varied the baseline value by a factor of 10; however, if that result was outside plausible or physical bounds, the working group lowered that range so it was plausible. Baseline values were selected from a variety of sources including RESRAD defaults and 1996 parameter values and were adjusted on occasion to ensure the physical range of interest was covered by a factor of three. Certain parameters were adjusted at later dates based on scientific or site-specific information. In some cases, the current values lie outside the range tested.

Table IV-2 lists the input parameters used as starting or "baseline" values for performing the sensitivity analysis. The actual parameter input values may differ somewhat from these baseline values, and may also differ among scenarios, but the range of inputs examined in the sensitivity analysis encompasses all of the values used as inputs to the various modeled scenarios.

Table IV-2. Sensitivity Analysis of Input Parameters

RESRAD 6.0 INPUT PARAMETERS	RESRAD 6.0 Default	Sensitivity Baseline Value	Range for Sensitivity Analysis		Value Used
			MINIMUM	MAXIMUM	
Contaminated Zone Parameters					
Area of Contaminated Zone (m^2)	10000	**5000**	100	250000	1400000
Thickness of Contaminated Zone (m)	2	**0.05**	0.01	0.25	0.15
Occupancy, Inhalation, and External Gamma Data					
Inhalation Rate (m^3/yr)	8400	**7000**	2448	19950	distribution
Mass Loading for Inhalation (ug/m^3)	0.0001	**0.00005**	0.00001	0.00025	distribution
Indoor Dust Inhalation Shielding Factor (unit less)	0.4	**0.8**	0.6	1	0.7
External Gamma Shielding Factor (unit less)	0.7	**0.8**	0.6	1	0.4
Indoor Time Fraction (unit less)	0.5	**0.68**	0.49	0.95	distribution
Outdoor Time Fraction (unit less)	0.25	**0.07**	0.02	0.25	distribution
Cover and Contaminated Zone Hydrological Data					
Density of Contaminated Zone (g/cc)	1.5	**1.6**	1.1	2.4	1.8
Average Annual Wind Speed (m/s)	2	**4.25**	3.04	5.95	4.2
Precipitation (m/y)	1	**0.381**	0.191	0.762	0.381
Ingestion Pathway, Dietary Data					
Fruit, Vegetable and Grain Consumption (kg/y)	160	**40.1**	13.4	120.3	distribution
Leafy Vegetable Consumption (kg/y)	14	**2.6**	0.9	7.8	distribution
Soil Ingestion (g/y)	36.5	**50**	25	100	36.5
Contaminated Fraction, Plant Food	-1	**0.5**	0.25	1	1
Ingestion Pathway, Nondietary Data					
Mass Loading for Foliar Deposition (g/m^3)	0.0001	**0.00005**	0.00001	0.00025	distribution
Depth of Soil Mixing Layer (m)	0.15	**0.05**	0.01	0.25	0.15
Depth of Roots (m)	0.9	**0.2**	0.05	0.8	0.15

Sensitivity Interpretation

The sensitivity analysis is centered around an initial input value for each parameter. The initial input parameters, or baseline values, were selected from values used in the 1996 RSAL analysis, except in cases where new information or new model requirements drove changes. Baseline values were reviewed prior to performing the analysis to ensure the baseline value and the resulting range of variability on that value were physically plausible and were compatible with the computational capabilities of the models.

Interpretation of the sensitivity analysis requires either a quantitative or a systematic qualitative ranking method to deal with the sensitivity outputs from RESRAD or Standard Risk Assessment Methodologies. The inputs and outputs were combined in a manner that first normalized the changes in input and output against baseline values so that a direct comparison of the relative changes would be possible. The necessity of this step can be made clear by considering that some inputs may have varied by amounts as small as 0.0004 units of measure, while others may have varied by 4900 units, yet the relative change is the same, say a factor of three. Without normalization to the baseline parameter values, their relative effects on dose could be lost to their disparity in magnitude.

Normalized responses have been calculated using three different algorithms; two are based on changes relative to the baseline, and the third is based on the range between the extremes of the dose calculation corresponding to minimum and maximum of the input range. The normalized responses are expressed as "sensitivity coefficients." The sensitivity coefficients are unit-less quantities and are calculated as follows:

Using the change between baseline input and minimum input:

$$S_{base\text{-}min} = (D_{base} - D_{min})/D_{base} \quad / \quad (P_{base} - P_{min})/P_{base}$$

where S, D and P denote Sensitivity Coefficient, Dose and Parameter respectively for minimum (min), baseline (base) or maximum (max, below) parameter inputs.

Using the change between baseline input and maximum input:

$$S_{base\text{-}max} = (D_{max} - D_{base})/D_{base} \quad / \quad (P_{max} - P_{base})/P_{base}$$

Using the change between maximum and minimum input:

$$S_{max\text{-}min} = (D_{max} - D_{min})/D_{base} \quad / \quad (P_{max} - P_{min})/P_{base}$$

In all three cases the resulting sensitivity coefficients were converted to absolute values (positive numbers) before the qualitative ranked comparison could be performed. This was done so that positive and negative changes of correspondingly equal magnitude would receive equal weighting in the final analysis. The sensitivity coefficients for each pathway and parameter were then sorted from highest to lowest, and "natural breakpoints" sought qualitatively in the resulting tabulation. Those parameters exhibiting the greatest contribution to changes in sensitivity coefficients were easily discriminated without further numerical analysis. This result can be seen in the example Pareto diagrams shown as Figures IV-2 and IV-3 for inhalation and soil ingestion pathways, respectively. Similar diagrams resulted for all the pathways examined.

Figure IV-2 Sensitivity Ranking - Inhalation, Pu-239, max-min basis, Top 20

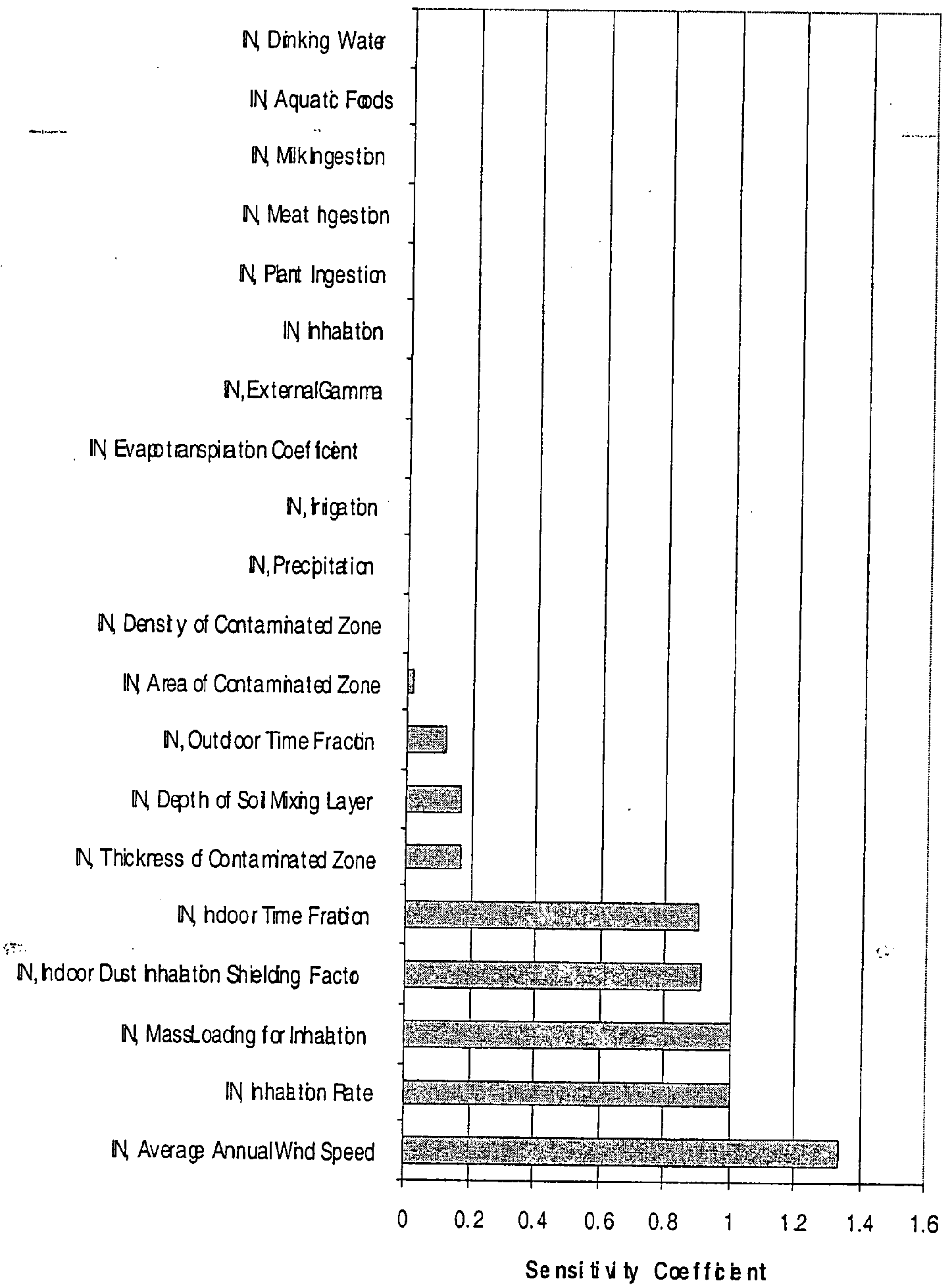

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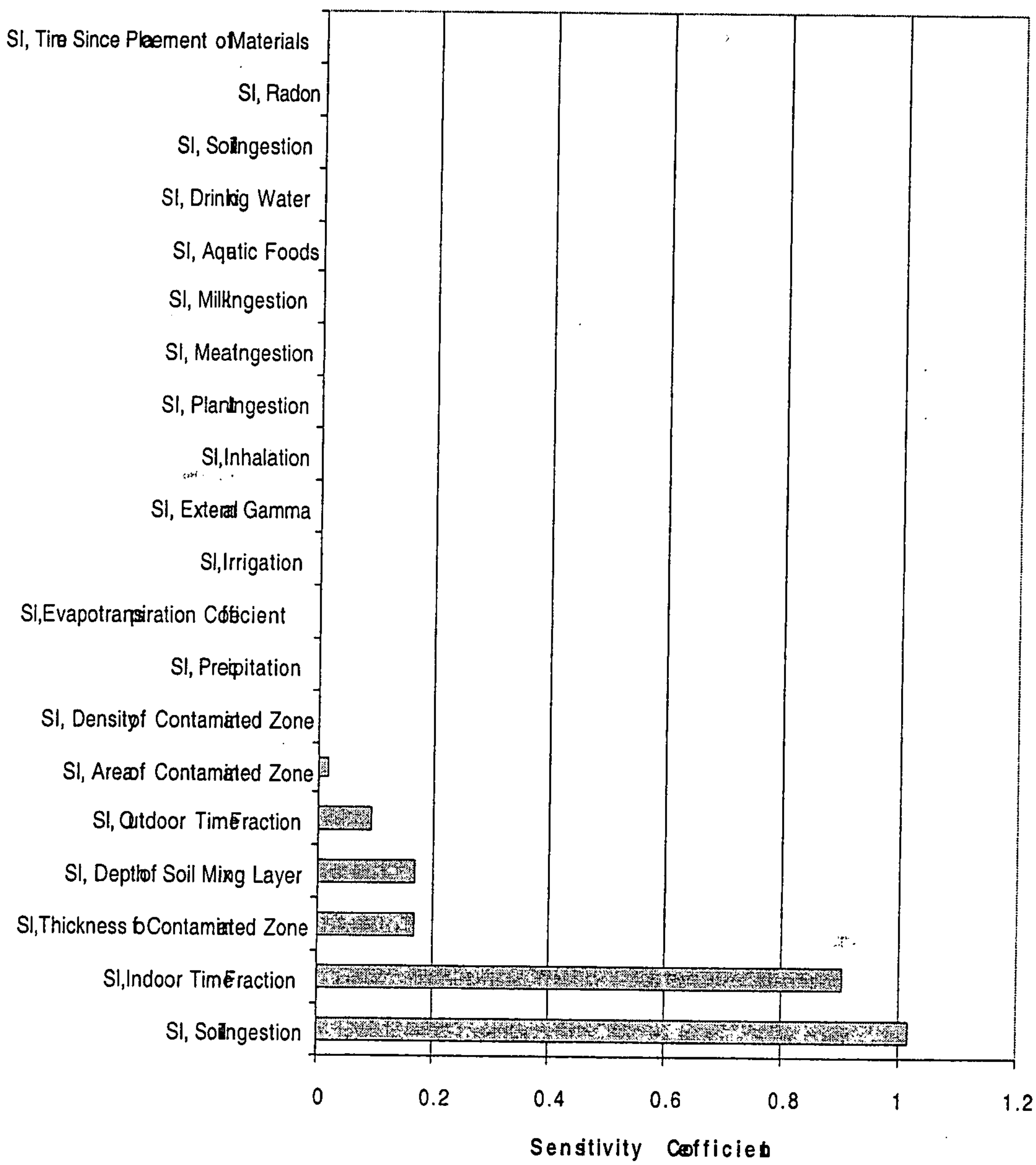
Figure IV-3 Sensitivity Ranking - Soil Ingestion, Pu-239, max-min basis, Top 20



The most sensitive parameters for a scenario are those parameters within a given pathway that will have the greatest influence or impact on the RESRAD (or Standard Risk Assessment Methodologies) model outputs. These figures show the ranked sensitivity coefficients representing $S_{max\text{-}min}$. As can be noted, only the first several coefficients (from the bottom) have values approaching one; that is, display changes in dose that are similar in relative magnitude to the change in parameter. Another less sensitive group displays a measurable change but notably smaller than that displayed by the first group; the remainder (including those not shown) display even smaller responses, suggesting that relatively large uncertainty in their selection would be rather inconsequential to the final result. The more sensitive parameters, however, need to be selected with great care if the final result is to represent the true consequences associated with exposure in the land-use scenario that is being investigated. The other sensitivity calculations, $S_{base\text{-}min}$ and $S_{base\text{-}max}$ did not prove as useful for assessing sensitivity itself, but provided insight into the mechanisms that might be causing the parameter to display a certain response. These observations are discussed in the next section.

The most sensitive parameters, determined from the combined analysis of all pathways for weapons-grade plutonium, are easily identified in Figure IV-4. The working group added "mass loading for inhalation" to this most sensitive list, because of the great interest in the post-fire scenarios, which could not be realistically tested using the sensitivity analysis protocols as defined by the RESRAD code. The most sensitive parameters were:

- Indoor Time Fraction
- Soil Ingestion Rate
- Mass Loading for Inhalation

Moderately sensitive parameters make up the remainder of the sensitive parameter list. They are:

- Thickness of the Contaminated Zone
- Depth of Soil Mixing Layer
- Depth of Roots
- Contaminated Fraction, Plant Food
- Fruit, Vegetable and Grain Consumption
- Outdoor Time Fraction
- External Gamma Shielding Factor
- Density of Contaminated Zone
- Average Annual Wind Speed
- Inhalation Rate and
- Indoor Dust Inhalation Shielding Factor

Moderately sensitive parameters are distinguished from sensitive parameters only by their reduced sensitivity response. Parameters having no sensitivity response are not listed or shown.

Pathway Sensitivity

The combined sensitivity analysis, when all active pathways are turned on at the same time, yields slightly different sensitivity parameter results than when the pathways are turned on separately. This is because of the additive influences of the different pathways to dose, even though some of the parameters only influence a single pathway. The reader should also be aware that the relative sensitivities would also be somewhat different if other factors influencing the calculations are changed.

The greatest influence on the relative contributions of the different pathways, given similar exposures among the pathways, would be the DCFs used to convert the exposure (amount of activity available to cause a health effect) into a dose (the measure of potential health effect). DSFs change when more becomes known about the mechanisms that cause health effects from exposure to radiation, or when more becomes known about the mechanisms that cause the material to be introduced into the body. For the analyses done here, the selection of dose conversion factors were chosen consistent with the most recently published values in International Commission on Radiation Protection (ICRP) Publications 60 through 72. The selection of dose conversion factors contained in these publications tend to attribute a higher dose conversion rate to the ingestion pathway for plutonium and americium than had been previously accepted, and a lesser rate to the inhalation pathway. This causes some differences in the partitioning of dose among the various pathways, as calculated by RESRAD.

As an example of changes that occurred, ICRP 72 modifies a number of the tissue weighting factors, adding significantly to the ingestion pathway by adding components for the colon, esophagus and stomach to the resultant selection of dose conversion factors. On the other hand, the lung mechanics are much better understood now, resulting in a reduction in the attribution of dose through this pathway.

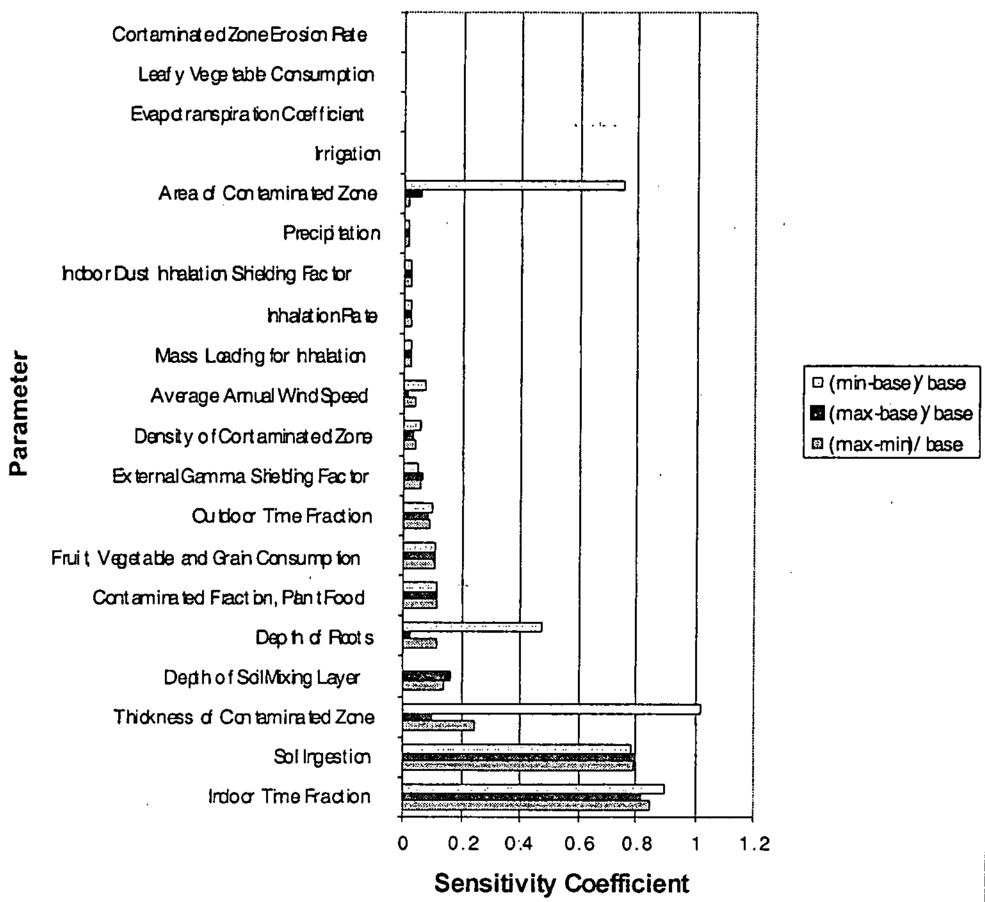
If the sensitivity analyses were to be repeated using selection of dose conversion factors previously published in ICRP 30, the results would be somewhat different, favoring parameters in the inhalation pathway more than is seen in the analysis presented here. However, the working group has examined the relative changes in these parameters and has concluded that the parameters being examined in detail would not have changed.

IV-3) Parameter Sensitivity

The working group focused on the sensitive and moderately sensitive parameters in its attempt to provide the most realistic and complete information possible. Both adult and child users have been considered where appropriate. The working group did review and discuss the selection of the less sensitive parameters, but only to the extent necessary to ensure completeness in the analytical process.

As mentioned above, some parameters displayed much more sensitivity than others. The working group sought to understand this behavior before final selection of parameter inputs so that anomalous results could be identified, if present. Again a graphical presentation of the sensitivity coefficients proved useful for identifying possibly anomalous results. Figure IV-5 displays a combined output of all three sensitivity coefficients. In these results it is possible to see examples where one or another of the three coefficients differs significantly from the others. These kinds of results could be indicative of unexpected non-linear behavior or behavior that suggests the parameter is interacting with other parameters. These individual parameters are discussed in the next section.

Figure IV-5 Sensitivity Coefficient Comparison - All pathways, WGPu, All non-zero values



IV-3A) Sensitivity Of Selected Parameters

Individual Sensitive Parameters are discussed below. In these discussions, related parameters are discussed together, even though they may not have similar sensitivities and consequently do not appear in the same order in Figure IV-5.

Indoor Time Fraction – The indoor time fraction has an important role in several of the exposure calculations, specifically inhalation and external exposure. In both cases, the exposure is reduced linearly with increased indoor occupancy, keeping all other factors constant. This factor becomes even more important when one considers that an increased indoor time fraction must be accompanied by a reduced outdoor time fraction in typical work-place or residential scenarios.

Outdoor Time Fraction – Outdoor time fraction does not display the same high sensitivity as the indoor time fraction. The outdoor time fraction is a linear factor in all of the pathways. The correlation with indoor exposure reduces the overall influence of this parameter in most scenarios.

Soil Ingestion – Soil ingestion rate is a very important parameter. The ingestion pathway has the greatest influence on dose and risk in the rural resident scenario. That dose and risk is linearly correlated with the soil ingestion rate.

Thickness of Contaminated Zone – The thickness of the contaminated zone has some influence on external exposure to gamma radiation, but its greatest influence is coupled with the influence of the "depth of roots" parameter. When the contaminated zone is very thin, and the roots extend significantly into uncontaminated soil, the dose and risk contribution from root uptake is dramatically reduced; conversely, when the contaminated zone is very thick, the roots are totally exposed to contamination and have the greatest uptake. Combined together, this sensitivity response can be non-linear as is displayed in the graphic.

Depth of Roots – Parallel discussion to "thickness of contaminated zone" discussion, above. The working group chose to make the depth of roots equal to the thickness of the contaminated zone, thus maximizing the potential uptake by roots.

Depth of Soil Mixing Layer – The depth of the soil mixing layer can be an important parameter in the inhalation pathway. This parameter is used to determine what depth within the contaminated zone is actually available for resuspension. Its sensitivity is mainly an artifact resulting from the baseline choice for the thickness of the contaminated zone. The working group chose to make the mixing layer depth equal to the thickness of the contaminated zone, maximizing the availability of contaminated material for resuspension.

Contaminated Fraction, Plant Food – Again, the ingestion pathway is the most significant pathway in the rural resident scenario, with a dose and risk that responds linearly with the availability of contaminated food material.

Fruit, Vegetable and Grain Consumption – In the important ingestion pathway, the rate of food consumption is linearly related to the calculated dose and risk through that pathway.

External Gamma Shielding Factor – The external pathway is not an important contributor to dose and risk, however that response is directly related to the amount of shielding that the rural resident enjoys during their significantly greater time spent indoors than outdoors.

Density of Contaminated Zone – This parameter has a non-linear influence on the external pathway, due to its role in attenuation of the gamma radiation coming from depth in the contaminated layer of soil. It will interact with the "thickness of the contaminated zone" parameter, discussed earlier. The density of soils at Rocky Flats is not highly variable, and the dose and risk from external radiation is not a large contributor. This parameter selection will have little influence on the modeled results.

Annual Average Wind Speed – The annual average wind speed parameter directly influences the concentration of radionuclides suspended in the atmosphere and available for inhalation. The parameter is non-linear with greatest changes evident at lower wind speeds. The annual average wind speed at Rocky Flats is a well-characterized and relatively constant quantity.

Inhalation Rate – Inhalation rate is linearly related to the dose and risk obtained through the inhalation pathway, an important pathway for all scenarios.

Indoor Dust Inhalation Shielding Factor – Inhalation is an important pathway. This parameter is most important to the rural resident and office worker scenarios because of the greater time spent indoors in these scenarios; it plays a similar but lesser role in the rural resident scenario. Shielding is treated as a linear factor, reducing the dose and risk that would be received were the receptor exposed to the mass loading outdoors.

The Area of the Contaminated Zone – This parameter is important to both the inhalation exposure pathway and the external exposure pathway. The radioactive contamination in the air is determined by a relationship between this contaminated surface area and "mass loading for inhalation." The working group chose a contaminated area large enough to saturate this pathway; that is, to cause its influence to be as great as possible. This chosen area is consistent with the actual area of contamination potentially subject to cleanup as a result of this potential RSAL analysis.

Mass loading for inhalation – The airborne concentration of inhalable particles (PM-10) in the vicinity of Rocky Flats is well characterized, varying from about 9. 4 micrograms per cubic meter ($\mu g/m^3$) to a high of about 16.6 $\mu g/m^3$, with a median of around 11.6 $\mu g/m^3$, based on the five most recent years of available PM-10 data from CDPHE. While this is a well-characterized distribution, it does not adequately represent potential perturbations to the annual mass loading that might be experienced by a future user at Rocky Flats. For example, more frequent routine soil disturbances, or increased wind erosion as the aftermath of a wildfire that denudes vegetation from large expanses of the soil surface would not be represented in the existing data. In this circumstance, other information must be sought to extend the observations to conditions for which there are no site-specific data. Since such estimates cannot possibly result in a single value that is known with precision, and because the range of possible values would be quite large, the mass loading for inhalation can be best represented by a probabilistic distribution of values, estimated from extrapolation of available data to represent possible future site conditions.

IV-4) Description of the Process for the Development of Probabilistic Distributions

As described previously, a sensitivity analysis was performed to identify the variables within each exposure pathway, which most strongly influenced the RSAL output. Those variables are summarized in Section IV-3). Following the conceptual approach shown in Figure IV-6, the RSAL working group evaluated the existing data to determine if a probability distribution could be developed for any or all of these influential variables. The existing data can be either site-specific or it can be surrogate data from EPA guidance documents, regional surveys, or the open literature.

Figure IV-6 Conceptual Approach for Developing Probability Distributions

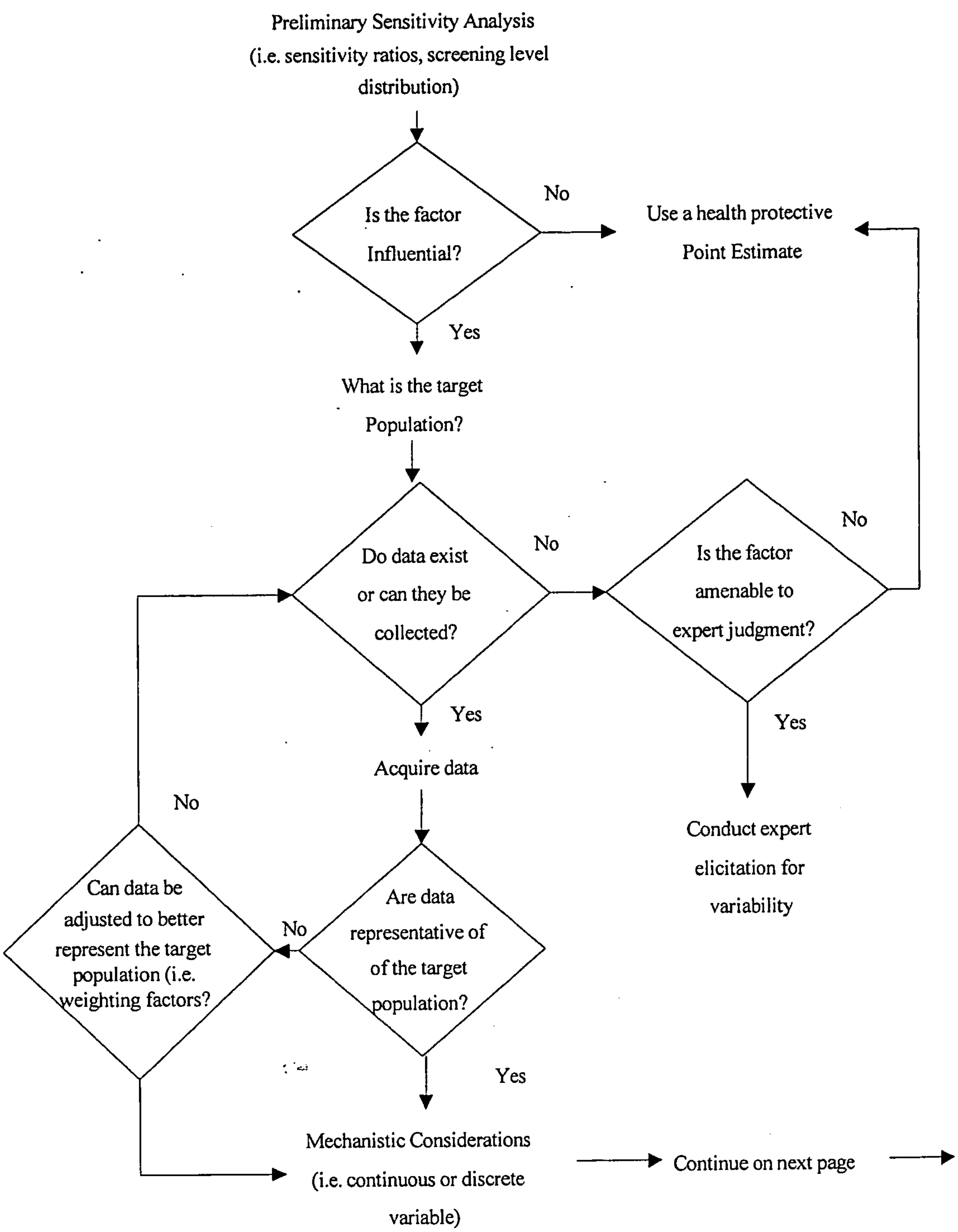
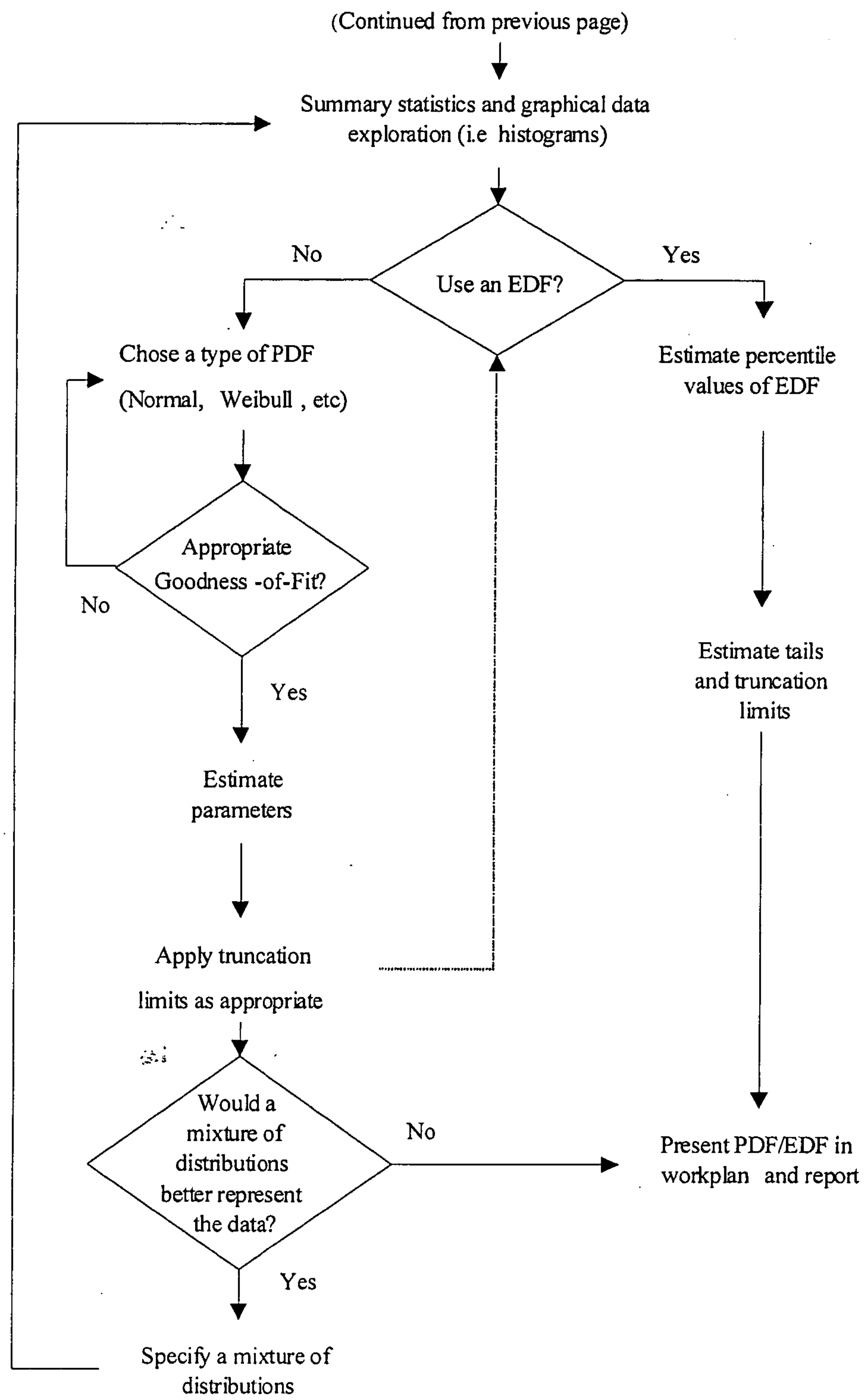


Figure IV-6 Conceptual Approach for Developing Probability Distributions
(continued from previous page)



For the majority of variables, such as exposure duration, soil intake rates, and body weight, site-specific data will not be available. Regardless of whether a data set comes from site-specific measurements or is obtained from published literature, it must be carefully evaluated for applicability to the target population at the site. The data set should either be from the target population or from a surrogate population, which is representative of the target population at the site. For example, daily intake rates of produce from an urbanized city in the northeast U.S. may not be representative of produce intake in a more rural western U.S. It would be far preferable to use data sets from western regions to represent residents near Rocky Flats, which was done. Questions to consider when evaluating the representativeness of a data set include: what are the populations of interest; how, when, and where are those populations exposed; the types of activities the populations engage in; the overall quality of the data design and collection, etc. The EPA's *Report of the Workshop on Selecting Input Distributions for Probabilistic* Assessments (U.S. EPA, 1999) is a good source for additional information on evaluating representativeness of data sets to a target population.

If, after considerable evaluation, the RSAL working group felt that the existing data were not adequate for developing a probability distribution, a health protective point estimate was selected instead. As a rule, the point estimate selected represented a reasonably maximally exposed or high end exposed individual. For example, the working group felt that the existing data on soil intake rates in adults was inadequate to develop a distribution for the wildlife refuge worker and for rural adult residents. As a result, EPA's recommended reasonably maximally exposed adult soil ingestion rate for adults in an agricultural setting (U.S. EPA 1991) was used for the wildlife refuge worker. If the working group determined that the existing data were adequate, then the next step was to fit a distribution to the data.

Sometimes more than one distribution may adequately characterize variability or uncertainty. In some cases, an empirical distribution function may be preferred over evaluating the fit of alternative probability models to a data set. The advantage of an empirical distribution function is that it provides a complete representation of the data with no loss of information and does not depend on the assumptions associated with estimating parameters for other probability models. The downside is that an empirical distribution function may not adequately represent the values at the extreme limits of a distribution due to limited sample size or poor sample design. Because EPA is required to develop human health preliminary remediation goals based on the reasonably maximally exposed individual, whom the limits of the distribution represents, this could become an important source of uncertainty. Another option might be to either extend the limits of the empirical distribution function or describe the data with an alternative model (e.g., probability density function). Graphical methods, goodness of fit tests, and examining the mechanistic basis of the biological or physical processes are all techniques that can be used to evaluate and select alternative probability distribution functions. It is not the intent of this report to describe these processes in detail, however EPA's *Risk Assessment Guidance for Superfund, Volume 3* (U.S. EPA, 2001) and the *Report of the Workshop on Selecting Input Distributions for Probabilistic Assessments* (U.S. EPA, 1999) are both useful sources of information on fitting and selecting distributions, and were used by the working group in developing distributions.

In Appendix A of this report, the process of selecting either a probability distribution or a point estimate for the most influential variables is discussed in detail. The data sets evaluated are presented as well as discussions pertaining to their representativeness and adequacy for developing probability distributions. If a distribution was developed for a given variable, Appendix A explains how the distribution was selected and fitted to the data.

IV 5) Selected Inputs for Sensitive Parameters

The results of the input selections for the most influential variables for both the rural residential and wildlife refuge workers are shown in Tables IV-3 and IV-4. If a probability distribution was developed for a variable, that distribution shape is described (e.g., lognormal, normal, etc.) and the statistical parameters, which define the distribution, are provided (e.g., mean, standard deviation, etc.). A very brief description of the data set from which the distribution was developed is provided in the comments field. For more detailed information on the data sets evaluated and the selection and fitting of the distribution the reader is referred to Appendix A.

Final Specific Input Values and Distributions for All Parameters

The input values for all of the parameters, including those specified as influential and those, which were, not, for all land use scenarios are shown in the spreadsheets in Appendices C and D.

A management decision was made to not develop probabilistic RSALs for the open space and office worker scenarios. These RSALS are based on a point approach only. The inputs to the variables for these two scenarios are shown in the spreadsheets in Appendix C.

Table IV-3 Summary of Exposure Variable PDFs for use in Rural Resident Scenario.

Exposure Variable	Population	Input Type		Input for RESRAD	Units	Input for STANDARD RISK ASSESSMENT METHODOLOGIES STANDARD RISK ASSESSMENT METHODOLOGIES	Units	Source and Comments
		Point	PDF					
Soil Ingestion Rate	Child		X	Bounded Lognormal-N (1.912, 1.371, 0, 365)	gm/year	Lognormal (47.5, 112, 0, 1000)	mg/day	Calabrese and Stanek (1997; 2000) and Stanek et al. (2001); Anaconda, MT (n=64), Best Linear Unbiased Predictor of 1-year average - empirical distribution function [{0.10, 0.25, 0.50, 0.75, 0.90, 0.95}, {2, 12, 25, 42, 75, 91}, 0, 150] mg/day; RESRAD unit conversion: g/yr = mg/day x 0.001 g/mg x 365 day/yr, then transform to ln(x) and calculate (mean of ln(x), stdev of ln(x)) [parameters of Lognormal-N]. Given uncertainty due to sample size, maximum was increased to 1000 mg/day using professional judgment.
	Adult	X		36.5		100		Calabrese et al. (1990) Amherst preliminary adult study (n = 6 subjects for 3 weeks); 4 tracers with best recoveries (Al, Si, Y, Zr) yielded min (> 0) of [1 -17 mg/day] and max of [99 - 216 mg/day], with individual means ranging [5 - 77 mg/day]. EFH (EPA, 1997) cites Best Tracer Methodology and plausible range of 30 - 100 mg/day, which is consistent with Superfund defaults of 50 mg/day (non-contact intensive) and 100 mg/day (reasonably maximally exposed). Use of point estimate equal to EPA's standard default for resident workers is professional judgment given scarcity of data (1 study, n = 6). RESRAD unit conversion: g/yr = mg/day x 0.001 g/mg x 365 day/yr.
	Age-adjusted		X	Bounded Lognormal-N (4.464, 0.246, 0, 365)		Lognormal (89.5, 22.4, 0, 100)		mixture distribution based on sum of child (lognormal) and adult (pt. est.) weighted by exposure duration (ED); values given assume EDc= 6 yrs and EDa = 24 yrs; could be entered with ED as a random variable
Indoor Time Fraction	All Ages	X		0.85	unit less	1235 minutes/day	unitless	EFH 1997 (Table 15-131), Residence Indoor Time for all ages: 75th percentiles = 1235 minutes indoors + 210 minutes outdoors = 1445, ~ 1440 minutes/day (24 hrs)

Table IV-3 Summary of Exposure Variable PDFs for use in Rural Resident Scenario.

Outdoor Time Fraction	All Ages	X		0.15	unit less	210 minutes/day	unit less	
Plant Ingestion Rate, Homegrown								
vegetables, seasonal	Child -total Child - leafy Child - non-leafy		X	Lognormal-N (0.782, 1.775) Lognormal-N (-1.122, 1.775) Lognormal-N (0.621, 1.775)	kg/yr	Lognormal (10.57, 50) Lognormal (1.57, 7.45) Lognormal (9.00, 42.55)	kg/yr	EFH 1997 (Table 13-33, West), Consumer only Intake of Homegrown Vegetables, Seasonally adjusted (g/kg-day), unit conversions: kg/yr = g/kg-day x mean body weight (15 kg) x 0.001 kg/g x 350 day/yr, empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.01, 0.10, 0.20, 0.60, 2.58, 7.67, 15.70, 26.46, 46.78}, 0, 58.8]; fit to Lognormal. Leafy = 14.9%; Non-leafy = 85.1%.
	Adult - total Adult - leafy Adult - non-leafy		X	Lognormal-N (2.322, 1.783) Lognormal-N (0.418, 1.783) Lognormal-N (2.161, 1.783)	kg/yr	Lognormal (50, 240) Lognormal (7.45, 35.76) Lognormal (42.55, 204.24)	kg/yr	same as child, but for mean body weight = 70 kg, empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.04, 0.47, 0.94, 2.79, 12.05, 35.77, 73.26, 123.48, 218.30}, 0, 274.4]; fit to Lognormal. Leafy = 14.9%; Non-leafy = 85.1%.
	Age-adjusted Total Leafy Non-Leafy		X	Lognormal-N (2.221, 1.75) Lognormal-N (0.304, 1.75) Lognormal-N (2.035, 1.75)	kg/yr	Lognormal (43, 196) Lognormal (6.3, 28.6) Lognormal (35.8, 163.6)	kg/yr	mixed distribution based on sum of child and adult weighted by exposure duration (ED); values given assume EDc= 6 yrs and EDa = 24 yrs; could be entered with ED as a random variable. Leafy = 14.9%; Non-leafy = 85.1%.
total fruit, seasonal	Child		X	NA		Lognormal (12.2, 37.3)	kg/yr	EFH 1997 (Table 13-33, West), Consumer only Intake of Homegrown Fruit, Seasonally adjusted (g/kg-day), unit conversions: kg/yr = g/kg-day x mean body weight (15 kg) x 0.001 kg/g x 350 day/yr, empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.00, 0.30, 0.46, 1.51, 3.61, 9.50, 24.94, 44.84, 76.13}, 0, 96.6]; fit to Lognormal
	Adult		X	NA		Lognormal (57, 174)	kg/yr	same as child, but for mean body weight = 70 kg, empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.01, 1.39, 2.16, 7.03, 16.86, 44.35, 116.38, 209.23, 355.25}, 0, 450.8]; fit to Lognormal
	Age-adjusted		X	NA		Lognormal (48, 119)	kg/yr	mixed distribution based on sum of child and adult weighted by exposure duration (ED); values given assume EDc= 6 yrs and EDa = 24 yrs; could be entered with ED as a random variable

Table IV-3 Summary of Exposure Variable PDFs for use in Rural Resident Scenario.

total grain	Child		X	NA		**Lognormal (23.65, 26.4)**	kg/yr	EFH 1997 (Table 12-1, West), Per Capita Intake of Total Grain Including Mixtures, not Seasonally Adjusted; empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.00, 3.6, 5.9, 10.1, 16.4, 26.4, 41.9, 57.2, 102.4}, 0, 135.9];
	Adult		X	NA		**Lognormal (110, 123)**	kg/yr	same as child, but for mean body weight = 70 kg; empirical distribution function [{0.01, 0.05, 0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.99}, {0.00, 16.9, 27.7, 47.0, 76.7, 123.2, 195.5, 267.1, 477.8}, 0, 634.3]; fit to Lognormal
	Age-adjusted		X	**Lognormal (93, 98)**	kg/yr	NA		mixed distribution based on sum of child and adult weighted by exposure duration (ED); values given assume EDc= 6 yrs and EDa = 24 yrs; could be entered with ED as a random variable
fraction grain homegrown	All Ages	X		**0.01**	unit less	**0.01**	unit less	professional judgment that homegrown grain products will be minimal
Fraction produce that is leafy (Flp)	All Ages	X		**0.149**	unit less	**0.149**	unit less	EFH 1997, Table 9-21 for West
Non-leafy Veg + Fruit + Grain	Child		X	**Lognormal-N (2.024,1.042)**	kg/yr	**Lognormal (21.4, 56.6)**	kg/yr	Sum of Total Veg + Total Fruit + Total Grain x fraction HG (see above) = Log(43, 196) + Log(48, 119) + Log(93, 98)x0.01; assumes independence in ingestion rates; fit to lognormal PDF. Leafy = 14.9%; Non-leafy = 85.1%.
	Adult		X	**Lognormal-N (3.566, 1.446)**	kg/yr	**Lognormal (100.7, 268.3)**	kg/yr	Sum of Total Veg + Total Fruit + Total Grain x fraction HG (see above) = Log(43, 196) + Log(48, 119) + Log(93, 98)x0.01; assumes independence in ingestion rates; fit to lognormal PDF. Leafy = 14.9%; Non-leafy = 85.1%.
	Age-adjusted		X	**Lognormal-N (3.438, 1.416)**	kg/yr	**Lognormal (84.8, 214.9)**	kg/yr	Sum of Total Veg + Total Fruit + Total Grain x fraction HG (see above) = Log(43, 196) + Log(48, 119) + Log(93, 98)x0.01; assumes independence in ingestion rates; fit to lognormal PDF. Leafy = 14.9%; Non-leafy = 85.1%.
Inhalation Rate	Child		X	**Lognormal-N (8.084, 0.305)**	m^3/year	**Lognormal (9.3, 2.9)**	m^3/day	Based on Allan and Richardson (1998) and review of EFH recommendations
	Adult		X	**Lognormal-N (8.657, 0.237)**		**Lognormal (16.2, 3.9)**		Based on Allan and Richardson (1998) and review of EFH recommendations

Table IV-3 Summary of Exposure Variable PDFs for use in Rural Resident Scenario.

	Age-adjusted		X	Lognormal-N (8.573, 0.207)		Lognormal (14.8, 3.1)		assumes Edc = 6 yrs, Eda = 24 yrs
Occupancy Factor	All Ages	X		1.0	unit less	NA		Intake rates are specific to the resident, therefore, intake rates do not need to be adjusted.
Exposure Time (ET)	All Ages	X		NA		24.0	hrs/day	professional judgment that all of the potential exposure occurs during a full day
Exposure Frequency (EF)	All Ages		X	incorporated into time fractions indoors and outdoors		Triangular (175, 234, 350)	days/year	EPA CTE default of 234 d/yr based on EFH, 64% of time spent at home for men and women; truncation limits are professional judgment that max time is 7 days/wk x 50 wk/yr; minimum is 50% of max.
Exposure Duration (ED)*	All Ages		X	Bounded Lognormal-N (2.046, 0.988, 1, 87)	years	Truncated Lognormal (12.6, 16.2, 1, 87)	years	EFH 1997, Table 15-167, Residential Occupancy Period, empirical distribution function [{0.10, 0.25, 0.50, 0.75, 0.90, 0.95, 0.98, 0.99}, {2, 3, 9, 16, 26, 33, 41, 47}, 1, 87]; fit to lognormal
Mass Loading Factor (ML)	All Ages		X	empirical distribution function divided by 1000 - see notes	g/m^3	empirical distribution function - see notes	ug/m^3	empirical distribution function derived by Workgroup based on site-specific data and professional judgment. Units converted to ug/m^3. [{0, 20.2, 23.1, 50.7, 58.0, 95.7, 109.5, 200},{min, 0.338, 0.788, 0.919, 0.944, 0.969, 0.994, max}]
Indoor Time Fraction (F_{in})	All Ages	X		0.85	unit less	0.85	unit less	EFH 1997 (Tables 15-131 and 15-132), Minutes Spent Indoors and Outdoors (All populations), 75th percentiles: 1235 minutes indoors + 210 minutes outdoors = 1445, ~ 1440 minutes/day (24 hrs);
Outdoor Time Fraction (F_{out})	All Ages	X		0.15	unit less	0.15	unit less	
Indoor Dust Filtration Factor	All Ages	X		0.7	unit less	0.7	unit less	average of indoors (0.4) described in EPA Soil Screening Level Guidance for Radionuclides and Default in RESRAD, and outdoors (1.0); assumes resident will spend time indoors, where windows and doors will be open during summer months
External Gamma Shielding Factor (see comment)	All Ages	X		0.4	unit less	0.4	unit less	EPA 2000, Soil Screening Guidance for Radionuclides

* Exposure duration may be entered as a random variable is RESRAD 6.0; the set of input values for all exposure variables are determined for Year 1, and applied across all years throughout the exposure duration.

Table IV-4 Summary of Exposure Variable PDFs for use in Wildlife Refuge Worker scenario

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Table IV-4 Summary of Exposure Variable PDFs for use in Wildlife Refuge Worker scenario

Exposure Variable	Input Type Point	PDF	Input for RESRAD*	Units[1]	Input for STANDARD RISK ASSESSMENT METHODOLOGIESS TANDARD RISK ASSESSMENT METHODOLOGIES (units)	Units	Source and Comments
Soil Ingestion Rate (IRs)	X		36.5	gm/year	100	mg/day	Calabrese et al. (1990) Amherst preliminary adult study (n = 6 subjects for 3 weeks); 4 tracers with best recoveries (Al, Si, Y, Zr) yielded min (> 0) of [1 -17 mg/day] and max of [99 - 216 mg/day], with individual means ranging [5 - 77 mg/day]. EFH (EPA, 1997) cites Best Tracer Methodology and plausible range of 30 - 100 mg/day, which is consistent with Superfund defaults of 50 mg/day (non-contact intensive) and 100 mg/day (reasonably maximally exposed). Use of point estimate equal to EPA's standard default for resident workers is professional judgment given scarcity of data (1 study, n = 6). RESRAD unit conversion: g/yr = mg/day x 0.001 g/mg x 365 day/yr.
Inhalation Rate (IRa)		X	Beta (min, max, P, Q) = (8.8,16.0, 1.79, 3.06)	m^3/day	1.1 + (2.0 - 1.1) x Beta (1.79, 3.06)	m^3/hr	Insufficient data from EPA EFH to generate PDF of breathing rates; PDF generated by varying the weighting factors for light, medium, and heavy activity (1.1, 1.3, and 2.0 m^3/hr)- see Table B.2-14 of RMA report and CDPHE analysis (Diane Niedzwiecki); Best-fit for beta (chi-square = 0.175), shape parameters are given and yields values between 0 and 1.0; for Crystal Ball, modify for scale using: min + (max-min)x beta; for @Risk, modify for scale using: min + beta; unit conversion m3/day = m^3/hr x 8 hr/day.
Occupancy Factor	X		1.0	unit less	NA		Intake rates are specific to the Wildlife Refuge worker, therefore, intake rates do not need to be adjusted.
Exposure Time (ET)	X		NA		8.0	hrs/day	professional judgment that all of the potential exposure occurs during a full workday

Table IV-4 Summary of Exposure Variable PDFs for use in Wildlife Refuge Worker scenario

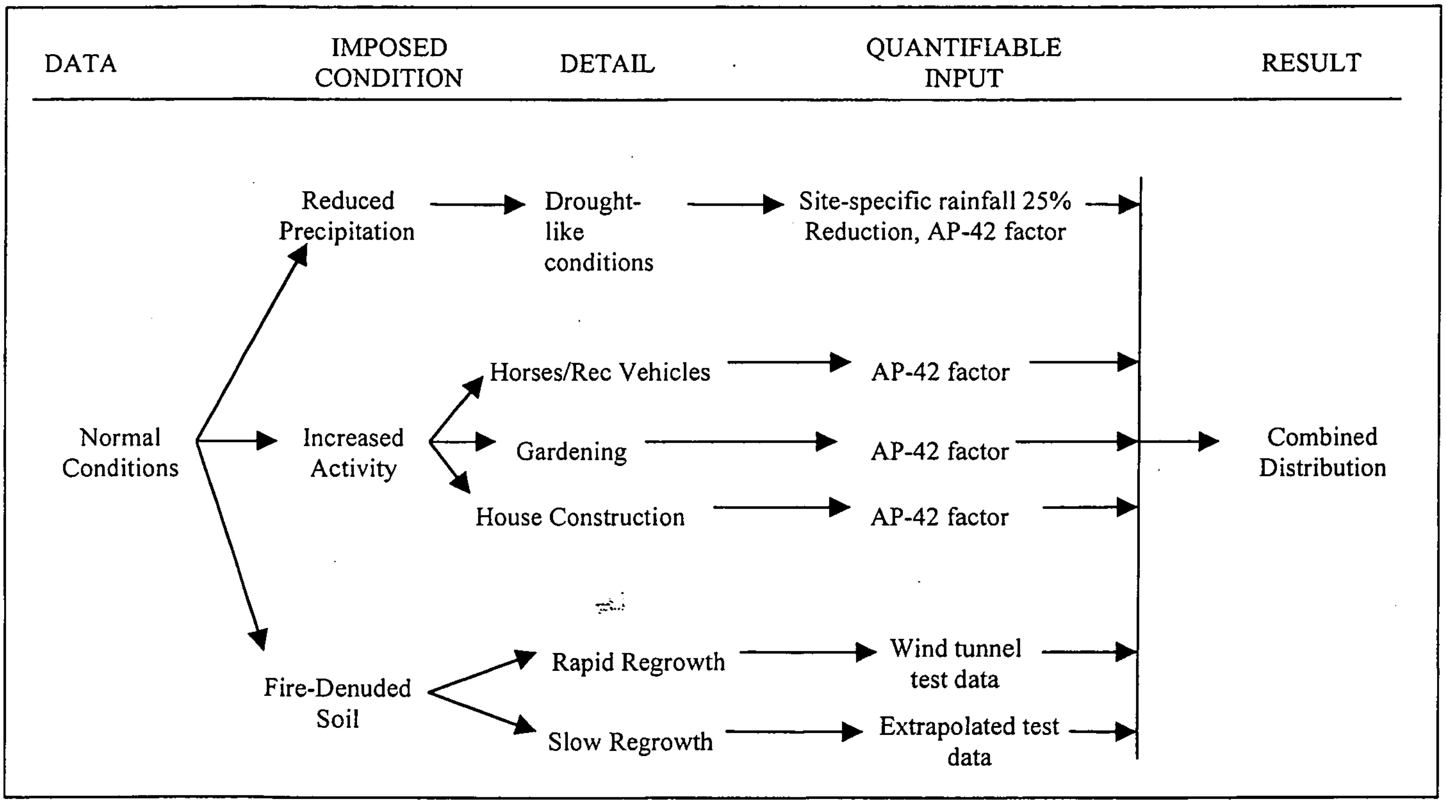
Exposure Frequency (EF)		X	NA		Truncated Normal (225, 10.23, 200, 250)	days/year	RMA report summarizing survey data for biological workers (n=20) (pp. B.3-149 - 150); truncation limits are professional judgment that minimum full time work is 4 days/wk x 50 wk/yr; max is 5 days/wk x 50 wk/yr.
Exposure Duration (ED)		X	Truncated Normal (7.18, 7, 0, 40)	years	Truncated Normal (7.18, 7, 0, 40)	years	RMA report summarizing survey data for biological workers (n = 20) (pp. B.3-172-175); truncation limits are professional judgment that values are nonnegative and within 5 SD's of the mean
Mass Loading Factor (ML)		X	empirical distribution function divided by 1000 - see notes	g/m^3	empirical distribution function - see notes	ug/m^3	empirical distribution function derived by Workgroup based on site-specific data and professional judgment. Units converted to ug/m^3. [{0, 20.2, 23.1, 50.7, 58.0, 95.7, 109.5, 200},{min, 0.338, 0.788, 0.919, 0.944, 0.969, 0.994, max}]
Indoor Time Fraction (F_{in})	X		0.5	unit less	0.5	unit less	RMA survey states ~ 0.5 time spent indoors
Outdoor Time Fraction (F_{out})	X		0.5	unit less	0.5	unit less	
Indoor Dust Filtration Factor	X		0.7	unit less	NA		average of indoors (0.4) described in EPA Soil Screening Level Guidance for Radionuclides and Default in RESRAD, and outdoors (1.0); assumes worker will spend time indoors, where windows and doors will be open during summer months
External Gamma Shielding Factor (see comment)	X		0.4	unit less	0.4	unit less	EPA 2000, Soil Screening Guidance for Radionuclides

* Exposure duation may be entered as a random variable in RESRAD 6.0; the set of input values for all exposure variables are determined for Year 1, and applied across all years throughout the exposure duration.

IV-6) Description of Problem Related to Mass Loading

In order to adequately describe the mass-loading parameter needed to describe future conditions, a conceptual model evolved as illustrated in Figure IV-7. The model presents several different conditions that might occur as a result of changes in land use. As a base case the working group considered present conditions at Rocky Flats. From that base condition, predictable effects of possible tilling and light recreational vehicle or horseback riding usage were considered. Such uses would be possible in all scenarios, to some extent, and were considered as a multiplier on the base case. The resulting modified mass loading will be referred to in this discussion as the "scenario mass loading." Other modifications to the scenario mass loading are driven by more specific events, such as periods of reduced rainfall (drought-like conditions) or periods following a fire during which the soil would erode more easily due to wind. These infrequent, but possibly significant occurrences were represented as random periodic modifications to the scenario mass loading. In other words, the resulting mass loading is to be represented as a probabilistic frequency distribution.

Figure IV-7 Conceptual Model: Mass Loading Influences



Two mass loading distributions are necessary for input into the RESRAD model, the first representing inhalable particulate matter and the second representing the particulate matter that is available for deposition onto plants. The first was derived based on site-specific and statewide PM-10 data; that is, data for air concentrations of particulate matter less than 10 micrometers aerodynamic diameter which are more easily admitted to the respiratory tract of humans. The second, total suspended particulate matter (TSP) can be derived from the first by assuming a direct correlation with PM-10, based on site-specific data. Studies of the mechanics of inhalation actually show that particles with aerodynamic diameters greater than about 2.5 micrometers are unlikely to reach the lower respiratory tract (Godish, 1991). For the particles that do get into the lower respiratory tract, an even smaller fraction is actually deposited in the lungs (Godish, 1991). Particles that do not reach the lungs will be either expelled or ingested.

<u>Description of data available</u>
The mass loading at Rocky Flats has been measured for a number of years. The most recent and probably most representative measurements of mass loading in the area around Rocky Flats are from CDPHE's five-station network surrounding the perimeter of Rocky Flats. Six years of PM-10 data are available (1995-2000) and have been used to depict the distribution of annual average mass loading at Rocky Flats (see Appendix F). The annually averaged data are described by a distribution whose range is from 9.4 $\mu g/m^3$ to 16.6 $\mu g/m^3$ with a median value of 11.6 $\mu g/m^3$. This mass loading may be compared to measurements of statewide PM-10 annually-averaged mass concentrations ranging from 6.7 $\mu g/m^3$ to 51.4 $\mu g/m^3$, with a median of 20.3 $\mu g/m^3$ (U.S. EPA 2001b) (see Appendix F). Clearly, the existing mass concentrations at Rocky Flats are among the lowest in the state. It is noted that the statewide data are likely to be somewhat biased to higher mass loading conditions, due to the siting criteria generally used for such monitoring stations. These siting criteria dictate that the stations be sited in areas more likely to experience air quality problems. Data from the CDPHE database for Rocky Flats also show that TSP can be linearly regressed against the PM-10 concentrations with a slope of approximately 2.5 (see Appendix F). This value of 2.5 was used as a direct multiplier to derive the TSP distribution used to characterize plant deposition from the PM 10 distribution.

Other information available
The literature offers a number of sources from which to build an estimate of mass loading. These sources can provide various mathematical factors that are descriptive of processes causing increased resuspension of soils due to various soil disturbance mechanisms. A well-documented source of such information is contained in background information provided for EPA's "Compilation of Air Pollutant Emission Factors" (AP-42)(U.S. EPA, 1995). In particular, its discussions related to the generation of fugitive dust, and the influence of precipitation on dust generation were especially pertinent (MRI, 1998). Also in AP-42 are descriptions of other dust generating activities that appear suitable as surrogates for future activities that might be observed at the site. Also, there is literature available through the National Drought Mitigation Center (NDMC, 1995) and through state resources relating the incidence of drought to the meteorological data that are available from site-specific measurement programs.

Finally, related to the fire-aftermath, the Site was able to conduct a wind-erosion study to develop site-specific measurements of erosion potential that could be used to estimate potential post-fire mass-loading increases on an annual basis. These results are presented in two reports. The first (MRI, 2001a) deals with the erosion potential and its changes with time. The second (MRI, 2001b) characterizes the relative concentrations of radionuclides observed in the soil and in the airborne eroded soil. Both are pertinent to the RESRAD calculations.

Quantification of Probabilistic Events

The probabilistic mass-loading distribution was built from four factors: the scenario mass loading as a baseline, the low-precipitation case, the spring-fire case and the fall-fire case.

First, the scenario mass loading was developed. Data relating the rate of emissions to soil-disturbing activities, suggest that the present-day mass loading at the Site could be expected to increase by as much as a factor of two (see Appendix F) due to moderate activities such as gardening, or use of light recreational vehicles or horses. While certainly coincidental, increases of this magnitude are consistent with the difference between the present 11.6 $\mu g/m^3$ median observed at the Site, and the state-wide median of 20.2 $\mu g/m^3$. The latter mass loading has been used as the scenario mass loading from which the probabilistic distribution was built.

A significant deficiency in rainfall can cause increased wind erosion of surface soil, even from vegetated areas. Site-specific data suggest that a reduction of 25% in annual rainfall, indicative of the onset of drought-like conditions (NDMC, 1995), occurs about 15 percent of the time, based on a data set spanning 37 years at the Site (see Appendix F). For purposes of developing a probabilistic distribution, the working group assumed that deficiencies in rainfall, to represent dryer than normal conditions, would influence about 25 percent of all modeled occurrences, a conservative assumption. The dust emission factor during such periods was adjusted upward about 14 percent based on guidance contained in AP-42 (MRI, 1998, p2-2). The calculation is simple -- for days with precipitation equal to at least 0.01 inches, fugitive dust is suppressed, and days with less than 0.01 inches of rain emit fugitive dust. The site-specific data were used to derive estimates of precipitation days in normal and dry years.

Data from wind-tunnel studies conducted after the 50-acre test burn at Rocky Flats in CY2000 provided estimates of erosion potential at different times following the grass fire. A spring-time fire on the site can be expected to cause an annual increase in erosion potential of about 2.5 times the potential without a fire (see Appendix F) due to removal of vegetation that provides a natural barrier to wind. In other words, after a spring-time fire, the annually-averaged mass loading should increase about 2.5 times. Within the next year or so, however, conditions would be expected to become normal. Extrapolation of these same data to a fire that might occur in the fall suggests that annual emissions would increase about 4.7 times, the fall timing presenting less favorable conditions for vegetative recovery. Based on the frequency of burns outlined in the Site's proposed controlled burn plan (DOE, June 2000) it has also been assumed that these fires could potentially involve a contaminated area once every 10 years. Half of those fires have been assumed to occur in the spring (warm seasons) when recovery is more rapid, and half have been assumed to occur in the fall (cold seasons), with slower recovery. This rate of fire occurrence is much greater than would be estimated for wildfires that might be caused by lightning or other causes, based on statewide data describing wildfire frequency (CO State Forest Service, 1999). Members of the working group also noted that controlled burns would not normally be prescribed in the fall, but such occurrences have been retained so as not to exclude wildfire events. The assumption of relatively frequent fall controlled-burn events constitutes a conservative assumption in the model.

Results

These probabilistic events were combined in a form that could be used by RESRAD and EPA's standard risk equations, specifically in the form of a discrete "continuous linear" (RESRAD's designation) distribution. The development of this distribution is detailed in Table IV-5. The eighth column in this table, labeled Grand Frequency, shows that the scenario mass-loading base conditions would be expected at Rocky Flats approximately 67.5 percent of the time, with dry-weather influencing this base condition about 22.5 percent of the time. Post-fire conditions, occurring in the upper 10 percent of the mass loading distribution are divided such that 90th to 95th percentile conditions are dominated by spring recovery events, including influence by dryer conditions, and 95th and greater percentiles are dominated by fall recovery events. The zero percentile and 100th percentile conditions, needed as input to the RESRAD model for this distribution, are represented by values of 10 $\mu g/m^3$ and 200 $\mu g/m^3$ respectively. The zero percentile is given as the low mass concentration observed in site-specific measurements and the 100th percentile is estimated based on the maximum value observed in the statewide PM-10 mass data, increased by a factor of about 4, weighted somewhat more heavily toward a possible fall-fire maximum value. [It should be noted here that the extremes of the distribution have little actual influence on the RESRAD or risk calculations, since the probability of such extreme occurrences is negligible.] However, actual ML value used for calculations was the 96th Percentile.

Table IV-5 Frequency Distribution Matrix; calculated cumulative frequency is shown in the two right-most columns. Zero and 100th percentiles are not shown.

Fire	Weight	Frequency	Precipitation	Weight	Frequency	Grand Weight	Grand Frequency	Mass Loading	Cumulative Frequency
No fire	1	0.9	Normal Precipitation	1	0.75	1	0.6750	20.2	0.338
No fire	1	0.9	Dry Conditions	0.14	0.25	1.14	0.2250	23.1	0.788
Spring fire	2.51	0.05	Normal	1	0.75	2.51	0.0375	50.7	0.919
Spring fire	2.51	0.05	Dry	0.14	0.25	2.87	0.0125	58.0	0.944
Fall fire	4.74	0.05	Normal	1	0.75	4.74	0.0375	95.7	0.969
Fall fire	4.74	0.05	Dry	0.14	0.25	5.42	0.0125	109.5	0.994

IV-7) Selection of Cancer Slope Factors

EPA classifies all radionuclides as Group A (known) carcinogens based on their property of emitting ionizing radiation and on extensive evidence from epidemiological studies of radiogenic cancers in humans (EPA, 2001b). At Superfund sites with radioactive contamination, EPA generally evaluates potential human health risks based on the radiotoxicity, i.e., adverse health effects caused by ionizing radiation, rather than on the chemical toxicity of each radionuclide present. An exception is uranium, where both radiotoxicity and chemical toxicity should be evaluated (EPA, 2001b). Usually only carcinogenic effects of radionuclides are considered, because in most cases, cancer occurs at lower doses than either mutagenesis or teratogenesis.

In order to evaluate the likelihood of cancer from exposure to individual radiogenic carcinogens, EPA's Office of Radiation and Indoor Air calculates cancer slope factor values for each individual radionuclide, based on its unique chemical, metabolic and radioactive properties. The cancer slope factors used in these risk calculations were obtained from Office of Radiation and

Indoor Air's most current (April 16, 2001) Health Effects Assessment Summary Tables (HEAST) and were, in large part, based on the risk coefficients derived in Federal Guidance Report No. 13, "Cancer Risk Coefficients for Environmental Exposure to Radionuclides" (EPA, 1999b). The only exceptions are the cancer slope factors for the soil ingestion pathway, which were not derived in Federal Guidance Report No.13. The cancer slope factors for the soil ingestion pathway were derived by Office of Radiation and Indoor Air in a parallel fashion to those presented in Federal Guidance Report No. 13 for the other pathways.

A cancer slope factor is an estimate of the probability of an individual developing cancer per unit intake of, or external exposure to a specific carcinogen over a lifetime. Inhalation and ingestion cancer slope factors for radionuclides are central estimates in a linear model of the age-averaged, lifetime radiation cancer risk for incidence of both fatal and nonfatal cancers per unit of activity ingested or inhaled. These cancer slope factors are expressed as risk/picocuries (EPA, 2001b). External exposure cancer slope factors for radionuclides are central estimates of the lifetime radiation cancer incidence risk for each year of exposure to external radiation from radionuclides distributed uniformly in a thick layer of soil. They are expressed as risk/year per picocuries/gram soil (EPA, 2001b). Thus, a cancer slope factor is similar to a dose conversion factor, but instead of assigning a unit dose for every unit of exposure (mrem/picocuries), a unit of risk is assigned for every unit of exposure (probability of adverse effect/unit radioactivity).

Cancer slope factors can be used to estimate lifetime cancer risks to members of the general population due to radionuclide exposures, when combined with site-specific media concentration data and appropriate exposure assumptions. The EPA Risk Assessment Methodology calculates the lifetime cancer risk associated with a radionuclide intake or external exposure as the product of the estimated lifetime intake, or external exposure to, a particular radionuclide and the radionuclide-specific cancer slope factor. This calculation presumes that risk is directly proportional to intake or exposure, i.e., it follows a linear, no-threshold model. Current scientific evidence does not rule out the possibility that risks from environmental exposure levels calculated this way may be over- or under-estimated. However, several recent expert panels (UNSCEAR, 1993, 1994; NRPB, 1993, NCRP, 1997) have concluded that the linear, no-threshold model is sufficiently consistent with the current understanding of carcinogenic effects of radiation that its use is scientifically justified for estimating risks from low doses of radiation. This linear, no-threshold model is universally used for assessing the risk from environmental exposure to relatively low environmental concentrations of radionuclides as well as to other carcinogens (below a risk of approximately 10^{-2}) (EPA, 1999b).

EPA has calculated cancer slope factors for most of the radionuclides and just as different radionuclides have different selection of dose conversion factors, different radionuclides generally have different slope factors. The slope factors also vary depending on route of exposure. Therefore, risk associated with inhaling 1,000 picocuries of uranium is different from that of inhaling 1,000 picocuries of cesium. Also, the risk associated with inhaling 1,000 picocuries of radium is different from that of ingesting 1,000 picocuries of radium via drinking water.

The radiation risk coefficients for cancer incidence which are the basis for the new cancer slope factors in HEAST incorporate the state-of-the-art models and methods developed in ICRP 60- 72 (EPA, 2001b). These new models take into account age and gender differences in radionuclide intake, metabolism, dosimetry, radiogenic risk, and competing causes of death. They are intended to apply to the general public who may be exposed to low-levels of radionuclides in the environment. These new risk coefficients incorporate:

- The most recent epidemiological evidence for cancer risk,
- Updated vital statistics from the 1989-91 U.S. decennial life tables, which define survival rates for an average person in the population,
- Improved biokenetic and dosimetry models from ICRP 60 - 72, which increase the predicted quantities for ingestion and decrease the predicted quantities for inhalation,
- More relevance to the general public. For internal doses, they incorporate age- and gender-specific absorbed dose rates, usage data and risk coefficients for specific cancer sites over the lifetime of the exposed population,
- Most recent external dosimetry (based on Federal Guidance Report No.12), which still is based on dose rates calculated for a reference adult male, applied to all ages and genders (EPA, 1993),
- The lung absorption type (M) and GI fractional absorption coefficient recommended by ICRP 71 for environmental exposures to plutonium and americium.

IV-8) Selection of Dose Conversion Factors

The RESRAD computer code requires the creation of and specification of a library of dose conversion factors, which is used for dose calculations. Separate values for dose per unit of radioactivity inhaled or ingested need to be specified for each isotope for which dose calculations are performed. Several isotopes of concern at Rocky Flats (notably the isotopes of plutonium) have different dose conversion factors depending on their behavior in the body (rate of absorption into the blood, rate of clearance from the lung, target organs, etc.), so decisions must be made as to which dose conversion factor to use.

The computation of dose conversion factors is fairly complicated, and requires the use of a separate model (outside the scope of RESRAD). The ICRP is a body of experts in all areas of the field of health physics which is tasked with developing and refining guidance on radiation protection, including the calculation of dose conversion factors for radioisotopes. The ICRP periodically reviews the experimental literature, updates its model assumptions about the way radioisotopes behave inside the body, revises its radiation protection guidance and/or revises the values of the dose conversion factors based upon the best available science at the time, and publishes their proceedings in numbered publications. The ICRP is recognized by all US regulatory agencies (NRC, DOE, EPA) as a highly credible source of radiation protection guidance.

ICRP originally created dose conversion factors for radioisotopes entering the body in its Publication 2 for worker exposure (ICRP, 1959), and there have been two comprehensive revisions since then. The first revision is captured in Publications 26 and 30 for worker exposure (ICRP, 1979). The second and most recent revisions take place in Publications 60 through 72 (1996) with compilations of dose conversion factors in Publication 68 (ICRP, 1994) for worker exposure and Publications 71 and 72 for exposure of the public (ICRP, 1995 and ICRP, 1996). Because of the timing of these revisions, the 1996 calculations of RSALs utilized the dose conversion factors from ICRP 30, and the Risk Assessment Corporation utilized the dose conversion factors from Publication 72. Since the later dose conversion factors are based upon a more complete research base, and are explicitly applicable to environmental exposure of the public, as opposed to radiation worker exposure, they are being used in the current calculations, and the RSAL working group decided to do so. The current NRC and Colorado radiation regulations relevant to determining total effective dose equivalents are based on ICRP 30.

There are several differences of note between the ICRP 30 and ICRP 72 approach. First, they are different biological models. Other significant changes include the development of dose conversion factors specific to various age groups; the revision of the lung model itself; a more extensive set of tissue weighting factors; and revisions to the ingestion dose conversion factor selections (specifically plutonium) to reflect the greater uncertainty inherent in environmental exposure to ingested radionuclides.

For the purpose of calculating the dose-based RSALs, the working group chose a relatively more rapid absorption type, the M type, for the behavior of Pu in the lungs. Note that the DOE and the EPA disagreed on this point (the DOE advocated use of the slowest absorption type, S type). All parties agreed, however, that while disagreement remained on the science and on the interpretation of the ICRPs, the calculation of RSALs was effected to only a minor extent and in the direction of greater conservatism.

TABLE IV-6: Comparative Inhalation selection of dose conversion factors (millirem per picocurie)

Isotope	ICRP 30 selection of dose conversion factors	ICRP 72 selection of dose conversion factors (adult)	ICRP 72 selection of dose conversion factors (child)
Plutonium 239/240	W 0.43	M 0.19**	M 0.29**
	Y 0.31*	S 0.06	S 0.14
Americium 241	W 0.44*	M 0.16**	M 0.26**
		S 0.06	S 0.15

*Value used in 1996
**Value to be used in 2001

TABLE IV-7: Comparative Ingestion selection of dose conversion factors (millirem/picocurie)

Isotope	ICRP 30 selection of dose conversion factors	ICRP 72 selection of dose conversion factors (adult)	ICRP 72 selection of dose conversion factors (child)
Plutonium 239/240	nitrates 0.0035	all forms 0.00093**	all forms 0.0016**
	all other 0.00037		
	oxides 0.000052*		
Americium 241	all forms 0.0036*	all forms 0.00074**	all forms 0.0014**

*Value used in 1996
**Value to be used in 2001

V. Risk and Dose Modeling Results and Discussion

The purpose of Section V is to tabulate the results from the risk and the dose calculations. Both the risk and dose results are calculated as RSALs for individual radionuclides. These results are presented in V-3 through V-7 below. For remediation purposes, the RSALS will be applied as sum-of-ratios wherever both plutonium and americium (its decay product) are present together in the environment. The approach for calculating sum-of-ratios is discussed in Section V-1 below, and the actual sum-of-ratio values are shown in Table V-1.

Table V-1. Dose and Risk Calculations for Plutonium in Surface Soil Adjusted by Sum-of-Ratios Method* (pCi/g)

Land Use Scenario	Risk Levels			25-mrem annual dose
	10^{-4}	10^{-5}	10^{-6}	
Wildlife refuge worker [a]	490	49	5	862
Rural Resident – adult [a]	173	18	2	209
Rural Resident– child [a]				244
Open Space User – adult [b]	1047	105	10	11797
Open Space User – child [b]				4842
Office Worker [b]	596	60	6	2289
Resident Rancher[b]	-	-	-	45

*This example accounts for additional activity from Am using a sum-of-ratios method, and assumes that the Am:Pu activity ratio equals 0.1527 and that only Am and Pu are present.
[a] Probabilistic (Percentile consistent with Reasonable Maximum Exposed individual)
[b] Deterministic
[c] RAC's original value was based on 15 mRem annual dose. This value is scaled to 25 mrem annual dose.

Table V-2. Dose and Risk Calculations for Americium in Surface Soil Adjusted by Sum-of-Ratios Method* (pCi/g)

Land Use Scenario	Risk Levels			25-mrem annual dose
	10^{-4}	10^{-5}	10^{-6}	
Wildlife refuge worker [a]	75	8	1	132
Rural Resident – adult [a]	26	3	0.3	32
Rural Resident– child [a]				37
Open Space User – adult [b]	160	16	2	1801
Open Space User – child [b]				739
Office Worker [b]	91	9	1	350
RAC Resident Rancher – adult	-	-	-	7[c]

* This example accounts for additional activity from Pu using a sum-of-ratios method, and assumes that the Am:Pu activity ratio equals 0.1527 and that only Am and Pu are present.
[a] Probabilistic (Percentile consistent with Reasonable Maximum Exposed individual)
[b] Deterministic
[c] RAC's original value was based on 15 mRem annual dose. This value is scaled to 25 mrem annual dose.

V-1) Dose Calculations for Each Scenario

If multiple radionuclides are present in the environment, the sum-of-ratios method is typically used to account for the contribution of each single isotope towards the dose or risk-based limit. Measured values of all radionuclides present are compared to action levels by dividing the measured value of each radionuclide by its respective action level, then adding the ratios. If the sum of the individual ratios is greater than one, then the limit is exceeded.

$$\frac{R1_M}{R1_{AL}} + \frac{R2_M}{R2_{AL}} + \frac{R3_M}{R3_{AL}} + \ldots < 1$$

where, $R1_M$ = measured value of the first radionuclide, etc

$R1_{AL}$ = action level of the first radionuclide, etc.

If the proportion of each radionuclide in the soil (activity ratio) is known, a derivation of this formula can be used to adjust the single radionuclide values to produce examples of levels that might be applied during remediation. The formula to derive a sum-of-ratios-adjusted action level for plutonium is:

$$Pu_{SR} = (Am_{AL})(Pu_{AL}) / (\_\ Pu_{AL} + Am_{AL})$$

where,
Pu_{SR} = sum-of-ratios-adjusted action level for plutonium
Pu_{AL} = action level for plutonium
Am_{AL} = action level for americium
_ = Am:Pu activity ratio

The sum-of-ratios-adjusted action level for americium can then be calculated by:

$$Am_{SR} = (1-(Pu_{SR} / Pu_{AL}))(Am_{AL})$$

Whenever a sum-of-ratios-adjusted action level is calculated, it is important that an actual measured americium:plutonium activity ratio be used. Using an actual activity ratio, examples of levels that might be applied in the field can be calculated. The 903 Pad Characterization Report (DOE, 2000) developed an americium:plutonium activity ratio of 0.1527 (at 10 pCi per gram of Am) based on a linear regression of data.

V-2) Risk Modeling Results for Each Scenario

The results of the risk-based RSALs are presented for the rural resident (Table V-3), the wildlife refuge worker (Table V-4), the office worker (Table V-5) and open space user (Table V-6). The RSALs for the office worker and open space user were estimated using a point estimate approach. Single values representing a reasonable maximum exposed individual were input to the equation and a single RSAL value was calculated for each radionuclide at the target cancer risk levels of 10^{-4}, 10^{-5}, and 10^{-6}. Using this table, an reasonable maximum exposed office worker who is exposed daily to 51 pCi per gram of Am-241 in soil over 25 years would have no greater than a 1 in 100,000 chance of developing cancer as a result of that exposure. Directly below each RSAL table is a table of percent contribution by exposure pathway. All of these exposure pathways were evaluated in the assessment and the RSALs are protective for cumulative exposure across all these pathways. All simulations are run with 10,000 iterations using Crystal Ball.

Table V-3. Risk Based Probabilistic RSALS for Individual Radionuclides for the Rural Resident

Radionuclide	Percentile*	Surface Soil Concentrations at Target Risk (pCi/g)		
		1E-04	1E-05	1E-06
Am-241	10th	135	13	1.3
	5th	87	9	0.9
	1st	37	4	0.4
Pu-239	10th	369	37	3.7
	5th	248	25	2.5
	1st	131	13	1.3

* 10th to 1st RSAL range corresponds to 90th to 99th RME risk range

Radionuclide	Average % Contribution by Pathway			
	Inhalation	Soil	Food	External
Am-241	7.7%	15.8%	28.5%	48.0%
Pu-239	29.8%	56.9%	11.9%	1.5%

The risk-based RSALS presented in Tables V-2 and V-3 for the rural resident and the wildlife refuge worker were estimated using a probabilistic approach. A range of values, described as probability distributions were input to the equations and the output is a range or distribution of RSALs which reflect variability in the population. A health-protective RSAL can then be selected from this distribution. The U.S. EPA is required by law to use the reasonable maximum exposed individual as a basis for evaluating human health risks and developing preliminary remediation goals (or RSALs) at Superfund sites (U.S. EPA 1990). In a point estimate approach the RSAL represents a soil concentration which is protective of the reasonable maximum exposed individual. In a probabilistic approach, EPA defines the 90-99th percentiles of a *risk* distribution as the recommended reasonable maximum exposed range, with the 95th percentile as the starting point for risk-decision making (U.S. EPA 2001a). Because RSAL calculations, for the most part, are the inverse of risk calculations, the reasonable maximum exposed range for RSALs corresponds to the 1st through 10th percentiles, with the 5th percentile as the recommended starting point. Similar to the point estimate approach, these probabilistic RSALs are presented at the target cancer risk levels of 10^{-4} to 10^{-6}. Using the recommended starting point of the 5th percentile, an reasonable maximum exposed resident exposed over a lifetime (both childhood and adult exposure) to 9 pCi per gram of Am-241 in soil would have no greater than a 1 in 100,000 chance of contracting cancer. This is in addition to the background cancer rate of approximately 1 in 3 in the U.S. (Colorado Central Cancer Registry, 1999). The percent contribution by exposure pathway is also shown for the resident and the wildlife refuge worker.

Table V-4. Risk-Based Probabilistic RSALS for Individual Radionuclides for Wildlife Refuge Worker

Radionuclide	Percentile*	Surface Soil Concentrations at Target Risk (pCi/g)		
		1E-04	1E-05	1E-06
Am-241	10th	351	35	3.5
	5th	306	31	3.1
	1st	243	24	2.4
Pu-239	10th	758	76	7.6
	5th	649	65	6.5
	1st	496	50	5.0

* 10th to 1st RSAL range corresponds to 90th to 99th RME risk range

Radionuclide	Average % Contribution by Pathway			
	Inhalation	Soil	Food	External
Am-241	7.1%	29.1%	0.0%	63.8%
Pu-239	17.4%	81.6%	0.0%	1.0%

Table V-5. Risk Based Deterministic RSALS for Individual Radionuclides for Office Worker (pCi/g)

Radionuclide	Surface Soil Concentrations at Target Risk (pCi/g)		
	10^{-4}	10^{-5}	10^{-6}
Am-241	511	51	5
Pu-239	725	73	7

Percent Contribution by Pathway

Radionuclide	Pathway		
	Inhalation	Soil	External
Am-241	22%	35%	43%
Pu-239	37%	63%	0%

Table V-6. Risk Based Deterministic RSALS for Individual Radionuclides for Open Space User (pCi/g)

Radionuclide	Surface Soil Concentrations at Target Risk (pCi/g)		
	10^{-4}	10^{-5}	10^{-6}
Am-241	955	96	9.6
Pu-239	1257	126	12.6

Percent Contribution by Pathway

Radionuclide	Pathway		
	Inhalation	Soil	External
Am-241	24%	37%	39%
Pu-239	37%	63%	0%

There are two points that are important to note when viewing these results. The first is that the estimates should not be viewed as exact. There are inherent uncertainties in the risk assessment process. The selection of future land use scenarios, risk or dose models, and parameter inputs all require careful evaluation of the existing information and an assessment of the strengths and weaknesses of that information. These strengths and weaknesses should be communicated to the risk decision makers for them to make health-protective remedial decisions. Decisions must be made using best professional judgment. Section VI provides greater detail on the uncertainties in this risk assessment process and the impact those uncertainties may have on the final results. As a general practice, the RSAL working group tried to present data as accurately and factually as possible without interjecting bias. However, when data sets were sparse or highly uncertain the working group defaulted to a conservative point estimate.

Another important point is that RSALs are initial guidelines and do not represent final cleanup or remediation levels. Risk managers must evaluate the remedial alternatives against the nine criteria described in the National Contingency Plan (U.S. EPA, 1990). These criteria are shown in Figure V-1 below. Achieving a target level of protection is one of the primary factors, but this objective needs to be balanced by criteria such as feasibility, permanence, state and community acceptance, and cost. A final cleanup level may differ from an RSAL following this comprehensive evaluation.

Figure V-1 Nine Criteria for Evaluation of Cleanup Alternatives

Threshold Criteria

1. **Overall protection of human health and the environment**
2. **Compliance with ARARs**

Balancing Criteria

3. **Long-term effectiveness and permanence**
4. **Reduction in toxicity, mobility, or volume through treatment**
5. **Short-term effectiveness**
6. **Implementability**
7. **Cost**

Modifying Criteria

8. **State acceptance**
9. **Community acceptance**

V-3) Dose Modeling Results

The results of the RESRAD dose calculations for single radionuclides are shown in Table V-7 below. The calculations were completed for rural resident adult and child, wildlife refuge worker, office worker and open space user, and the results are expressed in terms of single radionuclide surface soil activity concentrations that equate to a 25-mrem annual dose. RSALs for probabilistic calculations have been selected at the 95th percentile of the probability distribution.

Table V-7. Dose-Based RSALs for Individual Radionuclides

Land Use Scenario	^{239}Pu RSAL @ 25 mrem/y (pCi/g)	^{241}Am RSAL @ 25 mrem/y (pCi/g)
Wildlife Refuge Worker*	1000	951
Rural Resident – adult*	486	56
Rural Resident - child*	318	161
Open Space User – adult**	14640	9278
Open Space User – child**	5718	4824
Office Worker**	2722	2199

* Probabilistic (95th percentile of probability distribution)
** Deterministic

VI. Variability and Uncertainty in the Risk Assessment

As discussed in Section IV, probabilistic risk assessment uses probability distributions to characterize variability and uncertainty in risk estimates. If one or more of the equation inputs are distributions, the output of a probabilistic approach is a distribution of soil action levels. If the input distributions represent variability, then the output distribution can provide information on variability in risk in the population of concern.

One of the main goals of this probabilistic assessment was to try to determine the impact of variability in the most sensitive exposure parameters on the resulting risk, dose and RSAL calculations. Variability refers to true heterogeneity or diversity that occurs within a population or sample. For example, within a population that incidentally ingests soil from the same source and with the same contaminant concentration, the risks from that ingestion may vary. This may be due to differences in exposure (i.e., different people ingesting different amounts of soil, having different body weights, different exposure frequencies, and different exposure durations), as well as differences in response (e.g., genetic differences in resistance to a chemical dose, or physiological differences in amount of soil absorbed from the gastrointestinal tract). Differences among individuals in a population are referred to as inter-individual variability. Differences for one individual over time are referred to as intra-individual variability (EPA, 2001a).

The distributions used as inputs to the risk equations in this probabilistic assessment to calculate RSALS, for the most part, characterize the inter-individual variability inherent in each of the exposure assumptions, based on the currently available data. Thus, the impact of the natural heterogeneity in such variables as intake rates or exposure frequencies on the calculated risk or dose estimates or on the RSALs were evaluated quantitatively in this assessment. Compared to a simple point estimate calculation, such as was done in the 1996 RSALs calculations, this quantitative evaluation more completely and accurately characterizes the impact of the variability in the more important input parameters on the final risk, dose, and RSAL calculations.

The RSAL results presented in Section V, for the rural resident and the wildlife refuge worker show probabilistic RSAL values at the 10th, the 5th and the 1st percentile for target risks of one in 10,000 (10^{-4}), one in 100,000 (10^{-5}) and one in one million (10^{-6}). For a given risk level, these percentiles largely reflect the variance in the calculated RSAL distribution resulting from the variability for each of the component distributions.

In contrast, no attempt was made in this assessment to quantify uncertainty. Uncertainty occurs because of a lack of knowledge about parameters, models or scenarios. It is not the same as variability. Collecting more and better data, while variability is an inherent property of the particular population or dataset can often reduce uncertainty. Variability can be better characterized with more data, but it cannot be reduced or eliminated (EPA, 2001a). While variability can affect the precision of risk estimates, uncertainty can lead to inaccurate or biased estimates.

Uncertainty can be classified into three broad categories, as applied to risk estimates, according to EPA's Exposure Assessment Guidelines (EPA, 1992) and the Exposure Factors Handbook (EPA, 1997):

1) Parameter Uncertainty - lack of knowledge about values assigned to estimate parameters or variables in the risk, dose or RSAL equation. This type of uncertainty can occur in each step of the risk assessment process, from data collection and evaluation, to the assessment of exposure and toxicity. Sources of parameter uncertainty can include systematic errors or biases in the data collection process, imprecision in the analytical measurements, inferences made from a limited database when that database may or may not be representative of the variable under study, and extrapolation or the use of surrogate measures to represent the parameter of interest (EPA, 2001a). The use of the conservative value of 25 years to describe the exposure duration for the Office Worker receptor is an example of a variable with some parameter uncertainty. In the absence of knowledge about the specific types of occupations which could eventually work at an office park at Rocky Flats some time in the future, it was decided to use the conservative reasonably maximally exposed default of 25 years (EPA, 1991), even though most people likely would not work that long in one location.

2) Model Uncertainty - lack of knowledge about model structure or use; whether the mathematical models or equations used to calculate exposure and risk, toxicity, dose, mass loading factor, RSALs, etc. adequately describe the physical or biological processes of interest. All models are simplified, idealized representations of complicated physical or biological processes. They may not always adequately represent all aspects of the phenomena they are intended to approximate or may not always capture important relationships among input variables (EPA, 2001a). Sources of model uncertainty can occur when important variables are excluded, interactions between inputs are ignored, or surrogate variables different from the variable under study are used. An example of model uncertainty dealt with by the RSAL working group during this assessment is whether the ICRP equations used to calculate the ICRP 30 or the ICRP 72 Dose Conversion Factors more accurately describe how particulates are handled by the lung. It was decided that the newer lung model used in the ICRP 72 calculations more accurately described how various parts of the respiratory system are impacted by particulates and in turn how absorption takes place in the various regions, and therefore that the ICRP 72 Dose Conversion Factors should be the basis for the current dose calculations.

3) Scenario Uncertainty – lack of knowledge necessary to fully define exposure, particularly to potential receptors in the future. The choice of which receptors to use in an assessment necessarily requires professional judgment, and needs to take into account a variety of factors, including local population growth characteristics and current conditions, political, social and economic pressures, etc. In addition, describing a particular land use in the future, necessarily is uncertain. The RSAL working group attempted to be conservative in deciding which exposure pathways would likely be complete in the future. The group also used available site-specific information, such as the amount of water available in the perched, shallow hydrostratigraphic unit, in order to calculate as realistic risks, doses, and RSALs as possible. The RSAL working group decided to calculate risks, doses, and RSALs for a wildlife refuge worker to represent a likely on-site receptor within the next 50 to 100 years when institutional controls are still in place, and for a rural resident to represent a condition in the future when institutional controls no longer exist. There is scenario uncertainty intrinsic in all of these choices.

The amount of uncertainty inherent in each of the distributions for the most sensitive parameters for this RSAL assessment varies. In some cases, such as for the adult soil ingestion rate, the available data were so limited that a decision was made to simply use a conservative point estimate value instead of a distribution for the RSAL calculation. In other cases, such as exposure duration for a resident, quite a lot of confidence can be placed in the distribution chosen. Whenever possible, particular sources of uncertainty in this assessment have been identified and discussed in detail in Appendix A. However, in this assessment, neither a two-dimensional Monte Carlo assessment capable of quantifying the uncertainty in several of the input distributions for the exposure parameters at a time nor a series of one-dimensional Monte Carlo assessments assessing the impact of a single parameter at a time was done. Therefore, the uncertainty in these final cumulative risk and RSAL calculations is only addressed qualitatively at this point in time.

The following tables summarize the qualitative impact of the different sources of variability and uncertainty in this assessment.

Table VI-1 Summary of Qualitative Impacts for All Scenarios

Assumption	Predominant Variability or Uncertainty Considered in this assessment	Rationale	Source	Impact of assumption on risk estimates
Location on site: Buildings are assumed to be on contaminated soil	Scenario uncertainty	Unless current buildings are used, which is not part of the current Site plans, construction of new buildings would have to disturb the surface soil. A new building to house wildlife refuge workers would probably NOT be built on contaminated ground.	Professional judgment	Conservative assumption, resulting in likely overestimation of external irradiation exposure to the resident, the wildlife refuge worker, and the office worker
Receptor location: Both RESRAD and the risk calculations assume that location of the receptor for soil ingestion and external irradiation are not in the same place as for inhalation.	Model uncertainty	The receptor is in the most highly impacted area for each pathway.	Model default for RESRAD, ease of calculation, and professional judgment	Conservative assumption resulting in likely overestimation of soil ingestion and external irradiation pathways in order to saturate inhalation exposure pathway.
New Dose Conversion Factors and Cancer Slope Factors based on ICRP 60 through 72 and FGR 13 were used to calculate the RSALs.	Model uncertainty	The dose conversion factors from ICRP 71 (external irradiation) and 72 (ingestion and inhalation) and the most recent cancer slope factors incorporate the recommendations of Federal Guidance Report 13, which used state-of-the-art models and methods that take into account age- and gender-dependence of radionuclide intake, metabolism, dosimetry, radiogenic cancer risk, and competing risk, as well as updated vital statistics and baseline cancer mortality data.	Professional judgment to use the most recent, scientifically justifiable information possible.	Conservativeness dependent upon exposure pathway. Overall, the estimate of total radiogenic cancer risk attributable to uniform total-body exposure from low doses of low LET radiation has increased by approximately 11-13% from the previous estimates using ICRP 30 values, primarily due to changes in the baseline cancer mortality rates for the U.S. population.

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New Dose Conversion Factors from ICRP 60 through 72 attribute a higher dose conversion rate to the ingestion pathway for Pu and Am than previously accepted.	Model uncertainty	ICRP 72 modified a number of tissue weighting factors, adding significantly to the ingestion pathway by adding components for the colon, esophagus and stomach to the resultant dose conversion factor. This more complete absorption model was considered to be more complete, and a better basis for assessing ingestion by the public than the simpler model upon which ICRP 30 dose conversion factors was based.	Professional judgment; ICRP 60 through 72 (ICRP, 1991 through 1996).	More realistic and more conservative estimate of gastrointestinal absorption than given by ICRP 30 values.
ICRP 72 Lung Dose Conversion Factors result in a reduced attribution of dose through inhalation pathway compared to that with ICRP 30 dose conversion factors.	Model uncertainty	ICRP 72 Inhalation Dose Conversion Factors are based on a new lung model, which more accurately reflects lung mechanics (ICRP 66), and are applicable to environmental exposure to the public as opposed to radiation worker exposure.	Professional judgment that the ICRP 72 values reflected the most recent and applicable scientific understanding of lung function (ICRP 66)	More realistic and less conservative than ICRP 30 inhalation dose conversion factors.
ICRP 72 Dose Conversion Factors and HEAST, (2001) inhalation cancer slope factors (which incorporated FGR13 recommendations) used in the RSAL calculations assumed lung absorption Type M (medium particulate) for Pu.	Model uncertainty	ICRP 71 and 72 Guidance indicates that Type M better reflects the type of lung absorption expected of "low-fired" plutonium oxides found in the environment at Rocky Flats. Plutonium oxides attached to sub micron size particles such as dried ocean sediments and soil particles are more rapidly absorbed into the blood than larger particles of relatively pure plutonium dioxide (ICRP 71).	Professional judgment based on guidance from ICRP 26 through 30 and 68 through 72.	Reasonably conservative assumption that could result in either an over- or under-estimate of risk.
The inhalation pathway model in	Model uncertainty	Basic assumption in RESRAD area factor calculation is that receptor is at	RESRAD model default	Very Conservative assumption that likely over-estimates risk.

RESRAD assumes that the wind is constantly blowing in the direction of the receptor.		center of downwind edge of contaminated zone. As wind direction changes, the entire contaminated zone is always in front of the receptor.		
Risk equation directly inputs site-specific mass loading value. Assumption is made that all dust is radioactive.	Model uncertainty	No attempt was made to determine a contaminated fraction of dust. All surface soil contamination was assumed to be available for resuspension by wind.	EPA Soil Screening Guidance for Radionuclides (2000) default	Very Conservative assumption that likely over-estimates risk from this pathway.
Mass Loading for inhalation used the median of state-wide annual average PM10 measurements rather than site-wide annual average PM10 measurements as a seed value.	Model uncertainty	To account for the possibility of a wider range of soil-intrusive activities occurring on-site in the future than have occurred during the period of recent site-wide PM10 measurements.	Professional judgment	Conservative assumption that could result in an over-estimate of risk.
Mass loading for inhalation assumed a grass fire would occur every year on the contaminated portion of the site.	Model uncertainty	Taking both lightning caused and a regular prescribed fire schedule into account, it was assumed fires could occur somewhere every year on a parcel of land the size of Rocky Flats. Furthermore, fires could occur on any particular location at Rocky Flats every 10 years.	Professional judgment	Very conservative assumption given both local fire occurrence data for fires of any size (CO State Forest Service, 1999) and probable low fuel load on land burned the previous year. Likely to result in an over-estimate of risk.
Mass loading for inhalation takes soil moisture level into account.	Model uncertainty	Site-specific wind tunnel data indicates moisture level impacts soil dustiness.	Professional judgment	Realistic assumption that attempts to accurately portray site conditions. It could result in either over- or under-estimate of risk.
Risk equation does not take radioactive decay over time into	Model uncertainty	EPA requires risks be calculated using the standard risk equations from RAGS, or the Soil Screening	Professional judgment	Conservative assumption that is likely to over-estimate risks.

account.		Guidance. These equations do not calculate the impact of radioactive decay.		

Table VI-2: Summary of Qualitative Impacts for Rural Resident Scenario

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
Receptor location: Resident is assumed to spend 100% of his/her time on-site within the approximately 300 acres that is contaminated above 10 pCi/g.	Scenario uncertainty	This is the most prudent assumption, given the possibility that the location of the contamination may some day be forgotten. In reality, residential development could occur anywhere over the entire 6400 acres.	Professional judgment	Very conservative assumption that is likely to result in an over-estimation of risk.
Insufficient water for drinking or irrigation for a multi-family development of 5 acre tracts	Scenario uncertainty	Shallow aquifer will not support water uses of a development of this size. In addition, it is highly unlikely that a purchaser of a 5 acre ranchette would rely on a shallow well for water supply. Also opinion of Actinide Migration Panel is that plutonium is extremely insoluble, making migration by any pathway but particulate movement in water unlikely.	Site-specific data; professional judgment of Site and State water experts	Realistic assumption that attempts to accurately portray site conditions.
Adult soil ingestion will use EPA's reasonably maximally exposed default values as point estimates. Reasonably	Parameter uncertainty	Calabrese (1990) adult study, which has an n = 6 was not sufficient basis for developing a distribution, and was not designed to reflect soil ingestion by an adult; it was designed to calibrate tracer absorption from soil	EPA reasonably maximally exposed values are contained in U.S. EPA, (1991) Risk Assessment Guidance for Superfund, Supplemental	Relatively conservative estimate of higher end adult soil ingestion rate. It could result in an over- or under-estimate of risks.

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
maximally exposed = 100 mg/d		for the child dataset. EPA's reasonably maximally exposed values fell in the range shown by the Calabrese study, and given the paucity of other data, were considered to be a reasonable estimate of the higher end of adult soil ingestion based on the limited data available (Calabrese et al., 1989; Calabrese et al., 1990, Davis et al., 1990; Van Wijnen et al., 1990).	Guidance, "Standard Default Exposure Factors", Interim Final.	
Child soil ingestion will use Calabrese (1999, 2000) dataset from Anaconda study as basis for developing soil ingestion rate distribution. Lognormal (47.5,112,0,1000)	Variability	Best dataset available. Anaconda study occurred in western soils; soil was sieved so only particles smaller than 250 microns were assessed, reflecting dust that readily attaches to hands; more of the dataset was used, i.e., the outlier criteria did not eliminate as much of the dataset as that used in previous studies.	Calabrese, et al. (1999, 2000).	Realistic estimate for normal children in that occasional ingestion of up to 1000 mg/d is accounted for during the 6 years of more intensive hand-mouth activity of childhood. This estimate did not cover the subset of the population that exhibits significant pica behavior, and would under-estimate risks in that case.
The childhood soil ingestion PDF has multiple sources of uncertainty associated with it.	Parameter uncertainty	Professional judgment, including the following factors: determining trace element concentrations in non-soil sources; estimating GI-transit time from food to fecal samples; implementing	Stanek et al., 2001	As realistic estimate as currently possible. This estimate could over- or under-estimate risks.

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Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
		exclusion criteria to remove unreliable daily estimates for certain tracer elements; inconsistency among tracer elements; assuming that intra-individual variability is characterized by a lognormal distribution, and that all individuals exhibit the same intra-individual variability; selecting a maximum value for truncating the PDF that characterizes inter-individual variability.		
Home-grown meat, milk and eggs not considered	Scenario uncertainty	Pu and Am do not accumulate in meat or milk to any appreciable extent; pathway contributions were assumed to be minor, and were not calculated for this assessment, even though for some future residents these pathways potentially could be complete. In addition, the small, 5 acre plots assumed under this scenario could not supply all the forage or hay required for larger animals; outside feed would have to be supplied.	Smith and Black, 1975; Johnson, 1989; Kaiser Hill/RMRS (1996)	Realistic assumption that attempts to accurately portray likely behavior of plutonium and americium in animal products.
PDFs for homegrown produce	Variability	Best data available.	Professional judgment	Realistic estimate of average exposure. It could result in an

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
and grains reflect the variability in average ingestion rates.				over- or under-estimate of risks.
All of the home-grown produce is contaminated, i.e., contaminated fraction is set = 1.	Parameter uncertainty	Conservative assumption to cover situation when garden is in contaminated soil.	Professional judgment	Conservative assumption that could over-estimate risks if garden is not located in a contaminated area.
Seasonally-adjusted home-grown fruit and vegetable estimates for the West were used.	Variability	Best, most applicable datasets available. Seasonal variability for grains is probably a minor source of variability since grains may be eaten on a daily basis throughout the year. However, homegrown fruit and vegetable consumption is more likely to show seasonal variations.	U.S. EPA (1997) Exposure Factors Handbook	Realistic assumption that attempts to portray likely behavior of future residents. It could result in either an over- or under-estimate of risks.
Data from consumers only, rather than per capita data was included in the PDFs for home-grown produce.	Variability	Professional judgment that estimates for consumers only would be more representative.	U.S. EPA (1997) Exposure Factors Handbook	Realistic assumption that attempts to portray likely behavior of future residents. It could result in either an over- or under-estimate of risks.
Age-specific data was used to develop PDFs for home-grown produce.	Variability	Professional judgment that residential exposure would begin during childhood (< 7years) and continue through adulthood (> 7 years). Therefore, child ingestion rates and adult ingestion rates need to be accounted for.	U.S. EPA Exposure Factors Handbook (1997)	Realistic assumption that attempts to portray likely behavior of future residents. It could result in either an over- or under-estimate of risks.

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Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
Each individual consumes at the same percentile levels for each week of a season and each season of a year.	Scenario uncertainty	Professional judgment that short-term food diary data available from the EPA's Exposure Factors Handbook adequately reflects longer term ingestion rates.	U.S. EPA Exposure Factors Handbook (1997)	Simplifying assumption that could result in either an over- or under-estimate of risks.
Potential correlations for a given individual in their dietary preferences and choices of foods grown at home were not maintained in the dataset.	Parameter uncertainty	Dataset in the EPA's Exposure Factors Handbook (1997) did not maintain individuals' food preferences so correlations could be performed. Rather, data is presented as the sum of average ingestion rates for each commodity. Therefore, correlations could not be maintained.	U.S. EPA Exposure Factors Handbook (1997)	Limitation of available data that could result in either an over- or under-estimate of risks.
1% of the ingested grains were assumed to be home-grown	Parameter uncertainty	Professional judgment, knowing that not a large segment of the population grows its own food grains.	Professional judgment	Realistic assumption that attempts to portray likely behavior of future residents. It could result in either an over- or under-estimate of risks.
Recent root and foliar uptake values were used instead of default values.	Parameter uncertainty	Ward Whicker's recent data suggests that Pu and Am uptake into vegetables may be higher than reflected by the default Baes (1984) values; Whicker is considered the resident expert on plant uptake of Pu and Am.	Whicker, et al. (1999)	Whicker's Bv and Br values are up to an order of magnitude greater than the old default values. Since Bv and Br are directly proportional to risk, this increase had a significant conservative impact on the risk calculation. Whicker's values were derived based on sandy soil from the Savannah River site, not the clayey Rocky Flats soils.

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Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
<u>Depth of roots</u> in RESRAD calculation is set at 0.15 m, which means that plant roots will be entirely contained within the contamination zone; no fractionation of uptake.	Parameter uncertainty	Professional judgment, in order to be conservative, and cover all situations for plants with shallower root systems, such as many vegetable garden plants.	Professional judgment	Conservative assumption that could result in an over-estimation of risks if plant roots grow deeper than 0.15 m or if contamination zone is not as thick as 0.15 m (which is the case for the majority of plutonium and americium contamination at Rocky Flats).
<u>Contamination zone thickness:</u> Soil is assumed to be uniformly contaminated to a depth of 15 cm (0.15 m)	Parameter uncertainty	In order to accurately account for the possibility that all contaminated surface dust eventually can be inhaled. All surface soil profiles taken from Rocky Flats indicate that 90% of the contamination is in the upper 15 cm.	Professional judgment and RESRAD model assumption that a single contaminated soil depth is applicable to all pathways	Conservative, especially for all pathways other than inhalation, since any soil disturbance will result in mixing and dilution of the initial surface contamination. In most areas, the contamination is in the top 2-3 cm, and most garden vegetables have roots going deeper than that.
<u>Mass Loading for plant deposition</u> was calculated based on the Site observation that the TSP/PM10 ratio is about 2.5/1	Parameter uncertainty	Particles of all sizes could deposit on plants.	Professional judgment and site-specific information	Realistic assumption that attempts to accurately portray site conditions. It could result in an under- or over-estimate of risk.
<u>Inhalation Rate for adults</u> is a derived PDF, lognormal (16.2, 3.9) m3/day, (lognormal-N (8.657, 0.237), based on activity levels and	Variability	Existing time activity and breathing rate studies were reviewed and incorporated into probability density functions to describe minute volumes and times spent at various activity levels.	Professional judgment that the derived PDFs were the best available data to apply to risk assessments for North Americans	Realistic estimate of average exposure, which could result in an over- or under-estimate of risks.

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
minute volume data for adults aged 20-59 years over a 24 hour period.		Monte Carlo simulations were then run to estimate average 24 hour inhalation rates based upon these input probability density functions. This data better accounts for the variability within the available data than do older point estimates.	(Allan and Richardson, 1998; and EPA Exposure Factors Handbook, 1997).	
Inhalation Rate for a child is a derived PDF, lognormal (9.3, 2.9) m3/day, (lognormal-N (8.084, 0.305) m3/y) based on activity levels and minute volume data for normal healthy toddlers, ages 6 months to 4 years, over a 24 hour period.	Variability	Existing time activity and breathing rate studies were reviewed and incorporated into probability density functions to describe minute volumes and times spent at various activity levels. Monte Carlo simulations were then run to estimate 24 hour inhalation rates based upon these input probability density functions. This data better accounts for the variability within the available data than do older point estimates.	Professional judgment that the derived PDFs were the best available data to apply to risk assessments for North Americans (Allan and Richardson, 1998; and EPA Exposure Factor s Handbook, 1997).	Realistic estimate of average exposure. It could result in an over- or under-estimate of risks.
Inter-individual variability in inhalation rates at most activity levels are generally very low.	Variability	Key inhalation rate studies tend to report lognormal distributions fit to the available data; distributions are generally positively skewed, with more minute volumes nearer the lower end of the reported ranges. There	Professional judgment based on review of literature and apparent fit of available data to lognormal distribution shape by Crystal Ball (Allan	Realistic assumption that attempts to portray likely behavior of future residents. It could result in either an over- or under-estimate of risks.

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
		is remarkable consistency in estimates for both children and adults, with average inhalation rates among toddlers and young children exhibiting a range of about 1 m3/day (8.7-9.7 m^3/d) and adults exhibiting a range in average inhalation rates of about 6 m3/day (11.3-17.5 m^3/d)	and Richardson, 1998)	
The inhalation rates fit lognormal distributions.	Parameter uncertainty	The Allan and Richardson (1998) data that was the basis for the PDFs derived for this assessment provided graphical summaries of the fits, but no description of goodness-of-fit test statistics. All results are within the range of values recommended by U.S. EPA EFH for risk assessment (EPA, 1997).	Allan and Richardson (1998)	Limitation of available data. It could result in either an over- or under-estimate of risks.
Indoor Dust Filtration factor was set at 0.7, the average of the EPA recommended indoor value (0.4) and the outdoor value (1.0)	Parameter uncertainty	To account for the resident's spending some time indoors, with the windows open.	Professional judgment of RSAL working group	Conservative assumption that could over-estimate risks if windows are not routinely kept open during the warm months.
External Gamma Shielding factor set	Parameter uncertainty	EPA, Soil Screening Guidance for Radionuclides	EPA, 2000	Realistic assumption that attempts to accurately portray

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Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
at 0.4		(2000) default		amount of shielding provided by most buildings. It could result in an under- or over-estimate of risk.
Exposure Time assumes receptor is at home 24 hours/day when at home. This is equivalent to setting the Occupancy Factor in RESRAD = 1.0.	Parameter uncertainty	In order to conservatively cover situations where residents stay home most of the time, e.g., invalids. All of the exposure is assumed to occur at home.	Professional judgment	Conservative assumption that likely over-estimates risk if receptors do not stay at home 24 hours/day.
Indoor Time Fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.85.	Parameter uncertainty	An indoor time fraction of 0.85 (1235 minutes/d) is the 75^{th} percentile for the American population for the amount of time spent indoors/day at home.	EPA, Exposure Factors Handbook (Table 15-131) Minutes Spent Indoors, All Populations.	Relatively conservative assumption that attempts to portray likely behavior of future residents. It could result in either an under- or over-estimate of risks.
Outdoor Time Fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.15.	Parameter uncertainty	An outdoor time fraction of 0.15 is the 75^{th} percentile for the American population for the amount spent outdoors/day at home. (1.0 – 0.85 = 0.15)	EPA, Exposure Factors Handbook (Table 15-132) Minutes Spent Outdoors, All Populations.	Relatively conservative assumption that attempts to portray likely behavior of future residents. It could result in either an under- or over-estimate of risks.
Exposure Frequency is represented by a distribution based on reasonably conservative estimate	Variability	The central point in the triangular distribution is EPA's default central tendency recommendation for residential exposure, which	Professional judgment	Relatively conservative assumption that attempts to portray likely behavior of future residents. It could result in either an under- or over-estimate of

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Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk or dose estimates
of the number of days/year residents will stay home. Triangular distribution (175, 234, 350) days/year. This exposure frequency is incorporated into the indoor and outdoor time fractions for the RESRAD modeling.		was 64% of possible time spent at home for men and women. Truncated limits are professional judgment that the maximum time spent at home would be 7 d/wk x 50 wk/yr (i.e., a 2 wk/yr vacation) and the minimum is 50% of the maximum.		risks.
<u>Exposure duration</u> represented by a distribution intended to conservatively reflect actual U.S. residents' occupancy at a single location. Truncated lognormal (12.6, 16.2, 1, 87)	Variability	A probability distribution generated from the empirical distribution function reported by Johnson and Capel (1992) for n = 500,000 simulated individuals was fit to a lognormal distribution with a mean of 12.6 years and std dev of 16.2 yrs. This data includes all U.S. residents, from renters to people who live all their lives at one residence. The truncation limits reflect that range, and aimed for a plausible upper bound. There is relatively high confidence in this dataset and probability distribution.	Johnson and Capel, (1992) and EPA's Exposure Factors Handbook, 1997, Table 15-167, Residential Occupancy Period.	Conservative assumption that attempts to portray likely behavior of future residents. It could result in either an under- or over-estimate of risks.

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Table VI-3: Summary of Qualitative Impacts for Wildlife Refuge Worker Scenario

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk estimates
Time on-site: Worker is assumed to spend 100% of his/her time on-site within the approximately 300 acres that is contaminated above 10 pCi/g.	Scenario uncertainty	This is the most prudent assumption, given the possibility that the location of the contamination may some day be forgotten. In reality, the workers are likely to move over the entire 6400 acres, as well. Moreover, Institutional Controls will delineate locations of any remaining contaminated areas to this receptor in the near term.	Professional judgment	Very conservative assumption that is likely to result in an over-estimation of risk.
Wildlife Refuge Worker is assumed to be an adult, and children do not routinely spend significant portions of their time on-site; there is no on-site day care facility.	Scenario uncertainty	Day care facility is not consistent with purpose of the proposed Rocky Flats Wildlife Refuge	Professional judgment, consistent with the purposes of the Rocky Flats Refuge in the proposed legislation, the National Wildlife Refuge System Administration Act of 1966, as amended by the National Wildlife Refuge System Improvement Act of 1997 (16 USC 668dd-668ee), and the Fish and Wildlife Service Manual (Guidance) Section 603 FW2	Realistic assumption that attempts to portray likely behavior of future wildlife refuge workers. If children do spend a significant period of time on-site risks calculated in this assessment would likely be under-estimated.

Adult soil ingestion will use EPA's reasonably maximally exposed default values as point estimates. reasonably maximally exposed = 100 mg/d	Parameter uncertainty	Calabrese (1990) study, which has an n = 6 was not sufficient basis for developing a distribution, and was not designed to reflect soil ingestion by an adult; it was designed to calibrate tracer absorption from soil for the child dataset. EPA's reasonably maximally exposed values fell in the range shown by the Calabrese study, and given the paucity of other data, were considered to be a reasonable estimate of the higher end of adult soil ingestion.	EPA reasonably maximally exposed values are contained in U.S. EPA, (1991) Risk Assessment Guidance for Superfund, Supplemental Guidance, "Standard Default Exposure Factors", Interim Final.	Relatively conservative estimate of higher end adult soil ingestion rate. Could result in an over- or under-estimate of risks.
Adult Inhalation Rate distribution calculated based on survey results of 20 biological workers' reported work activities and average breathing rates for each of those activity levels. Best fit to a beta distribution, with shape characteristics, 1.1 +(2.0-1.1) x Beta (1.79, 3.06).	Variability	Survey of 20 biological workers at Wildlife Refuges who spent at least 50% of their time on-site, outside, performing a variety of activities was best source of information on the type of work Refuge workers were likely to perform. Short-term inhalation rate recommendations for outdoor workers are from EPA's EFH, which appeared to be the best, most justifiable source.	Refuge Worker activities and activity levels taken from survey performed for Rocky Mountain Arsenal, 1990. Breathing rate data from EPA, Exposure Factors Handbook, 1997.	Relatively conservative estimate of average inhalation rates for this type of worker, taking types of activities performed into account. Could result in an over- or under-estimate of risks.
Indoor Dust Filtration factor was set at 0.7, the	Parameter uncertainty	To account for the wildlife refuge workers spending some time indoors, with the	Professional judgment of RSAL working group	Conservative assumption that could over-estimate risks if windows are not routinely kept

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average of the EPA recommended indoor value (0.4) and the outdoor value (1.0)		windows open.		open during the warm months.
External gamma shielding factor set at 0.4	Parameter uncertainty	Soil Screening Guidance for Radionuclides default for use at sites with soil contaminated with radionuclides (EPA, 2000). The 0.4 value is recommended as appropriate for above-ground lightly constructed (wood frame) buildings.	EPA, (2000).	Conservative estimate that could over-estimate risks for heavily constructed buildings (block and brick).
Exposure time set at 8 hours/working day for full time work. This is equivalent to setting the Occupancy Factor for RESRAD = 1.0; all the exposure occurs during working hours, on-site.	Parameter uncertainty	Typical full time working day. This exposure time does not take over-time into account.	Professional judgment	Realistic estimate of average full time refuge worker's time spent on-site. It could over- or under-estimate risks if number of hours/day worked deviate significantly from 8 hrs/day.
Indoor Time Fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.5.	Parameter uncertainty	All exposure frequency and duration data were taken from the U.S. Fish and Wildlife Service survey of 20 biological refuge workers who spent at least 50% of their time onsite, outdoors.	Rocky Mountain Arsenal Proposed Final Integrated Endangerment Assessment/Risk Characterization, (1994) summarizes the survey of biological refuge workers.	The workers spend at least 50% of their time onsite, outdoors. Therefore, this fraction is the low end of the range. If future workers' time spent indoors varies from this 0.5 point estimate, their risks could be either over- or under-estimated.

Outdoor Time fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.5.	Parameter uncertainty	Outdoor time fraction is correlated with the indoor time fraction (1.0-0.5 = 0.5).	Professional judgment	The workers spend at least 50% of their time onsite, outdoors. Therefore, this fraction is the low end of the range. If future workers' time spent indoors varies from this 0.5 point estimate, their risks could be either over- or under-estimated.
Exposure frequency characterized by a truncated normal distribution (225, 10.23, 200, 250) days/year.	Variability	Data taken from survey of the subpopulation of wildlife refuge workers (n = 20), termed biological workers who spent at least 50% of their time on-site, outside (RMA, 1994). Relatively low number of workers in survey. Ranges of 200 d/y represent minimum full time work (4d/wk x 50 wk/y) to 250 d/y (usual full time work 5 d/wk x 50 wk/y). The maximum value of 250 d/y is consistent with the EPA's reasonably maximally exposed default. The lower bound of 200 d/y suggests that the range among different workers in the wildlife refuge survey was narrow. The arithmetic mean (225 d/y) is slightly greater than the average reported by the Bureau of Labor Statistics for all occupations (219 d/y). Overtime work is not included.	Professional judgment on truncation limits; Rocky Mountain Arsenal Proposed Final Integrated Endangerment Assessment/Risk Characterization, (1994) report summarizing survey results from biological refuge workers (n = 20) (pp. B.3-149-150), (RMA, 1994).	Realistic estimate of average full time refuge worker's time spent on-site. If future worker spends more or less time on-site, the risks could be under- or over-estimated.

<u>Exposure duration</u> characterized by a truncated normal distribution (7.18, 7, 0, 40) years.	Variability	Data taken from survey of 80 biological workers at wildlife refuges (RMA, 1990). Data included all workers, including those that did not spend a significant amount of time outside. Incomplete tenures were included. Ranges of 0 and 40 years were chosen to avoid negative values for exposure durations at a minimum, and so the maximum was approximately 5 standard deviations from the mean.	RMA report summarizing survey results from all wildlife refuge workers (n = 80) (pp. B.3-172-175), (RMA, 199).	Realistic estimate of average full time refuge worker's time spent on-site.

Table VI-4 Summary of Qualitative Impacts for Open Space User Scenario

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk estimates
Open space user is assumed to spend 100% of his/her time on-site within the approximately 300 acres that is contaminated above 10 pCi/g. .	Scenario uncertainty	This is the most prudent assumption, given the possibility that the location of the contamination may some day be forgotten. In reality, the open space users are likely to move over the entire 6400 acres, as well.	Professional judgment	Very conservative assumption that is likely to result in an over-estimate of risks.
Adults and children will be able to visit the wildlife refuge or open space.	Scenario uncertainty	Consistent with open space and wildlife refuge visitor usage reported by local Open Space agencies and parks.	Professional judgment based on Jefferson County Open Space survey, (1995); Boulder County Open Space survey, (1996).	Realistic assumption.
Adult soil ingestion rate will use _ of EPA's reasonably maximally exposed default value as a point estimate. Adult soil ingestion rate set at 50 mg/visit.	Parameter uncertainty	Open Space visitors would obtain only _ of EPA's default adult residential amount soil ingested while on-site. The other _ of the daily intake of soil could reasonably be expected to occur at other locations.	Professional judgment	Realistic assumption that could either result in an over- or under-estimate of risks.
Child soil ingestion rate will use _ of EPA's reasonably maximally exposed default value as a	Parameter uncertainty	Open Space visitors would obtain only _ of EPA's default child residential amount of soil ingested while on-site. The other _ of the	Professional judgment	Realistic assumption that could either result in an over- or under-estimate of risks.

point estimate. Child soil ingestion rate (for ages 1-6 yr) set at 100 mg/visit.		daily intake of soil could reasonably be expected to occur at other locations.		
Calculated Adult Inhalation Rate point estimate of 1.7 m3/hour.	Variability	Time-weighted average inhalation rate assuming _ light and 1/2 medium activity while hiking, and 1/3 light activity, 1/3 medium and 1/3 heavy while biking and jogging on-site. Short-term average inhalation rate recommendations for adults at different activity levels are from EPA's Exposure Factors Handbook, (1997) which appeared to be the best, most justifiable source.	Professional judgment on activities and activity levels based on Jefferson County Open Space Visitor Survey (1996). Breathing rate data from EPA, Exposure Factors Handbook, (1997).	Reasonably conservative estimate for an average exercising person, taking different activity levels into account. Could over- or under-estimate risks.
Indoor Dust Filtration factor was set at 1.0 since the receptor would spend all his time onsite outdoors.	Parameter uncertainty	To account for the Open Space or wildlife refuge visitors spending all of their time onsite, outdoors, hiking, biking, jogging, or performing some other type of exercise. Given current U.S. FWS funding, no visitor center is planned at Rocky Flats.	Professional judgment of RSAL working group	Realistic estimate for someone outside.
External gamma shielding factor set at 0.	Parameter uncertainty	EPA, (2000) Soil Screening Guidance for Radionuclides default.	EPA, (2000).	Realistic estimate for someone outside.
Exposure time set at	Parameter uncertainty	50th Percentile for the number	Professional	Realistic assumption that could

2.5 hours visiting the site/day		of hours/visit at Jefferson county Open Space parks.	judgment based on Jefferson County , Open Space Visitor Survey of several mountain parks (1996).	either result in an over- or under-estimate of risks.
Indoor Time Fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.	Parameter uncertainty	The Open Space receptor will spend all their time onsite, outdoors, since no visitor center is currently planned.	Professional judgment	Realistic estimate for someone outside.
Outdoor Time fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 1.0	Parameter uncertainty	Outdoor time fraction is correlated with the indoor time fraction (1.0-0 = 1.0).	Professional judgment	Realistic estimate for someone outside.
Exposure frequency is set at 100 days/year.	Parameter uncertainty	Jefferson County Open Space Visitor Survey (1996) 95th percentile of number of visits/year.	Professional judgment based on Jefferson County Open Space Visitor Survey, (1996).	Conservative estimate that could overestimate risks.
Exposure duration is set at 30 years.	Parameter uncertainty	Local residents could visit the park consistently over the entire time they live in area. EPA recommended reasonably maximally exposed Exposure Duration for residents is set at 30 years, the 90th percentile for Americans to live in one place.	EPA, (1991) Supplemental Guidance: Standard Default Exposure Factors and EPA, (1997) Exposure Factors Handbook.	Conservative estimate that could overestimate risks.

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Table VI-5 Summary of Qualitative Impacts for Office Worker Scenario

Assumption	Predominant Variability or uncertainty considered in this assessment	Rationale	Source	Impact of assumption on risk estimates
Worker is assumed to spend 100% of his/her time on-site within the approximately 300 acres that is contaminated above 10 pCi/g. .	Scenario uncertainty	This is the most prudent assumption, given the possibility that the location of the contamination may some day be forgotten. In reality, the office development could occur anywhere on the entire 6400 acres, as well.	Professional judgment	Very conservative assumption that is likely to result in an over-estimation of risk.
Office Worker is assumed to be an adult, and children do not routinely spend significant portions of their time on-site; there is no on-site day care facility.	Scenario uncertainty	This scenario was not expected to be the primary determinant of action levels, so it was not developed extensively.	Professional judgment, given limited resources and time.	Non-conservative assumption that could result in an under-estimation of risks if children do spend a significant portion of time on site.
Insufficient H_2O for drinking or irrigation for a commercial office development, which average about 30 acres in size in the Boulder, CO area.	Scenario uncertainty	Shallow aquifer will not support water uses of a development of this size.	Site-specific data; professional judgment of Site and State water experts	Realistic assumption that attempts to accurately portray site conditions.
Adult soil ingestion will use EPA's recommended central tendency default values as point estimates.	Parameter uncertainty	Calabrese (1990) study, which has an n = 6 was not sufficient basis for developing a distribution, and was not designed to reflect soil ingestion by an adult; it	EPA reasonably maximally exposed values are contained in U.S. EPA, (1991) Risk Assessment Guidance for	Realistic estimate of average soil ingestion rate expected by typical office workers.

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Central tendency value = 50 mg/d		was designed to calibrate tracer absorption from soil for the child dataset. EPA's default central tendency value fell in the range shown by the Calabrese study, and given the paucity of other data, were considered to be a reasonable estimate of average adult soil ingestion. Because office workers would not be expected to participate in soil contact intensive activities, the central tendency default value is a better estimate of likely soil ingestion rate.	Superfund, Supplemental Guidance, "Standard Default Exposure Factors", Interim Final.	
Adult inhalation rate for sedentary activities typical of office workers recommended by the ICRP 66 is used. Inhalation rate = 1.1 m3/hour	Parameter uncertainty	ICRP recommended inhalation rate for office workers is a conservative estimate of breathing rates for sedentary workers.	ICRP 66	Conservative estimate of average breathing rate for predominantly sedentary population.
Indoor Dust Filtration factor was set at 0.4, the EPA recommended indoor value (0.4).	Parameter uncertainty	Newer office buildings typically do not have windows that open; the buildings will be air conditioned.	Professional judgment of RSAL working group	Realistic
External gamma shielding factor set at 0.4	Parameter uncertainty	EPA, (2000) Soil Screening Guidance for Radionuclides default.	EPA, (2000).	Reasonably conservative estimate for slab construction.

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Exposure time set at 8 hours/working day for full time work. This is equivalent to setting the Occupancy Factor for RESRAD = 1.0; all the exposure occurs during working hours, on-site.	Parameter uncertainty	Typical full time working day. This exposure time does not take over-time into account.	Professional judgment	Realistic estimate of average full time refuge worker's time spent on-site.
Indoor Time Fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 1.0.	Parameter uncertainty	Any time spent outdoors would be minimal portion of day; e.g., walking to and from building parking lot.	Professional judgment	Realistic assumption that attempts to accurately portray likely site conditions.
Outdoor Time fraction for the RESRAD model and for risk inhalation and external irradiation calculation is set to 0.	Parameter uncertainty	Outdoor time fraction is correlated with the indoor time fraction.	Professional judgment	Realistic assumption that attempts to accurately portray typical office worker behavior.
Exposure frequency is set at 250 days/year.	Parameter uncertainty	EPA's Superfund guidance recommends assuming a worker will be at work 5 d/wk for 50 wks/yr, or 250 d/yr. This assumes 2 weeks vacation away from work per year. Overtime work is not included.	Professional judgment and EPA, default guidance (1991).	Realistic estimate of average full time worker's time spent on-site.
Exposure duration is set at 25 years.	Parameter uncertainty	Individual workers are assumed to work 25 years at	EPA, (1991) Supplemental	Conservative value used in absence of any specific

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		the same location. This is the 95th percentile for all U.S. workers, (U.S. Bureau of Labor Statistics, 1990)	Guidance: Standard Default Exposure Factors.	information on job titles.

Conclusions

It is apparent from the table that conservative assumptions were made at a number of points during this risk assessment in order to account for uncertainties in parameters, models and scenarios. As stated in Section IV, the RSAL working group generally tried to present data as accurately and factually as possible, without interjecting bias. However, when data sets were sparse or highly uncertain, the workgroup defaulted to a conservative position. Some of the most important conservatisms are intrinsic to the scenarios. For instance, the assumption that the receptors spend all of their time on the approximately 300 acres of the site that are currently most contaminated biases the results toward the conservative end. In addition, both the RESRAD model and the risk equations use conservative assumptions at several places that likely bias the results toward the conservative end. For instance, RESRAD's assumption that the wind is always blowing in the direction of the receptor, or the risk equations' lack of accounting for radioactive decay with time, both bias the results toward the conservative end. Thus, simply because the scenarios and the models used necessarily defaulted to conservative assumptions, the results of this assessment are not likely to under-estimate risks, doses or RSALs.

This conservatism is balanced somewhat by the use of average ingestion rates for specific populations when developing the various input parameter distributions. For example, datasets of average ingestion rates of produce and grains or average soil ingestion rates were used to develop the input parameter distributions. By doing this, it was hoped that overall, a balance could be struck that resulted in a reasonably conservative estimate of potential exposure. If an appropriate balance has been struck, the 95^{th} percentile risk or the 5^{th} percentile PRG recommended in EPA guidance as the starting point for choosing the PRG or in this case, the RSALs, may indeed truly represent the reasonably maximally exposed (EPA, 2001).

EPA draft probabilistic risk assessment guidance (EPA, 2001) recommends calculating a risk assessment using point estimates for comparison to the results of a probabilistic risk assessment. This has not yet been done using the assumptions decided upon by the RSALs working group for the 2001 assessment. The 1996 RSALs were calculated using point estimates, but with a different RESRAD model and several differences in assumptions. Therefore, it is not yet possible to completely compare the results of this probabilistic risk assessment to the results of a standard deterministic risk assessment estimate of an reasonably maximally exposed exposure.

Therefore, the RSAL working group has assumed that the 95^{th} percentile of risk results or the 5^{th} percentile RSALs calculated in this assessment correspond to the reasonably maximally exposed. The large number of scenario and model uncertainties for which conservative assumptions were made in this assessment do not support moving the reasonably maximally exposed estimate to a higher percentile of risk (lower of RSALs) within the reasonably maximally exposed range defined by the 90^{th} to about the 99^{th} percentiles of risk.

SECTION VII
DRAFT TASK 3 REPORT
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APPENDICES

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Appendix A - Justification and Supporting Documentation for the Input Parameters

This appendix documents the rationale for the selection of values that were used in performing RESRAD and risk model runs for the 2001 RSAL determinations.

Area of the Contaminated Zone

The RESRAD computer model performs two main calculations to assess the impacts of radionuclides in soil: 1.) a dose (or risk) calculation based upon soil concentrations of radionuclides which are input into the model (which could be thought of as the site conditions before cleanup), and 2.) an RSAL calculation which is based upon the inherent properties of the radionuclides identified as contaminants coupled with the other physical properties of the site (site conditions after cleanup to the RSAL value). In both cases the RESRAD model simplifies the calculation by assuming that the contamination is uniformly present throughout the area of the contaminated zone, which is an area in square meters (circular or other specified shape) presented as an input parameter. "Uniform" is taken to mean that a variation no greater than a factor of 3 times the mean value or less than 1/3 the mean value are present.

The assumption of uniform contamination is oversimplified when applied to a dose calculation at a site before cleanup, since the contamination is rarely uniformly distributed. (Performing multiple RESRAD runs on increments of the area of consideration, which are contaminated at different concentrations, and combining the results often addresses such a problem.) However, the assumption of uniform contamination is both reasonable and conservative when applied to the RSAL calculation, for a site after cleanup. Particularly, it is a conservative assumption, because, in assuming uniform contamination, it overestimates the actual situation (where some of the contaminated area has been cleaned up to below the RSAL value). Since the purpose of this Task is the computation of dose based and risk based RSALs, the use of the RESRAD model with this assumption should not give cause for concern.

The area of the contaminated zone has been identified as an important parameter in Section IV for the combined pathway sensitivity analysis. Inspection of the mathematical formulas used by RESRAD for each pathway (Yu et al., 2001,) shows that all pathways are independent of area, except the air inhalation and gamma exposure pathways. Moreover, work with the RESRAD gamma exposure pathway shows that it "saturates" at relatively small areas (less than 1000 m2 or about one fourth acre). This is understandable, since the exposure rate from gamma emitters drops off rapidly (inverse square law) with distance from the source.

The inhalation pathway, investigated alone, saturates relatively slowly due to the effect of the area of the contamination zone on the area dilution factor used by versions of RESRAD later than 4.65. When taken in combination with all other pathways, however, it is seen that the slow saturation of the inhalation pathway contributes very little to the total dose, which is dominated by soil and plant ingestion contributions (both area-independent). Selection of the value of 1,400,000 m^2 for the circular area of the contaminated zone (the area know to be contaminated above 10 pCi/g of plutonium at ROCKY FLATS), assures that the combined pathway analysis is based upon saturation conditions.

Density of Contaminated Zone

The density of the Contaminated Zone is 1.7 g/cm^3, which is the rounded average bulk density for the Rocky Flats Alluvium. The dry bulk density ($\_{_b}$) measurements summarized below are taken from the following reports:

- French Drain Geotechnical Investigation (EG&G, 1990)
- OU1 Phase III RFI/RI Report (DOE, 1994)
- OU4 IM/IRA Environmental Assessment Decision Document (DOE, 1994)
- OU2 Phase II RFI/RI Report (EG&G, 1995)
- Groundwater Recharge Study (EG&G, 1993)
- Geotechnical Engineering Study, Sewer Line Installation South of Central Avenue (Huntington, 1994)
- Geotechnical Engineering Investigation Report Addendum, Title III Waste Management Facility Design (Merrick & Co., 1995)
- Preliminary Conceptual Design Document for Sanitary Landfill (Merrick & Co., 1990)
- Geotechnical Investigation Report of OU5 (DOE, 1995)

Dry Bulk Density of Rocky Flats Alluvium

Number of Measurements	Average (g/cm^3)	Range (g/cm^3)	Standard Deviation
90	1.68	0.95 – 2.18	0.257

These measurements are from intervals deeper than the 15 cm depth of the Contaminated Zone and are therefore likely to be higher than densities typical of the Contaminated Zone. The denser the soil, the more activity per volume of soil and the greater the potential dose due to external irradiation.

Thickness of Contaminated Zone

More than 90% of the Pu-239/240 and Am-241 radioactivity measured in soil profiles for OU2 is contained in the upper 0.12 m, regardless of soil type or location. Near-surface physical activities (e.g., freeze-thaw cycles) and biological activities (e.g., earthworms and macropores along decayed root channels) are considered the most important factors in the vertical distribution of actinides at ROCKY FLATS. The thickness of this zone has been set at 0.15 m (6 in.), which corresponds to both the RFCA definition of surface soil and the default surface soil depth typically found in EPA guidance (EPA, 1992).

Depth of Roots

The depth of roots (d_r) is set at 0.15 m, equal to the thickness of the Contaminated Zone (T). The cover and depth factor for root uptake ($FCD_{p1}(t)$), therefore, is equal to 1 (no effect). If d_r is greater than T, a portion of the roots is outside the Contaminated Zone and the amount of root uptake would be fractionated by the ratio of the two intervals (T/d_r). This root depth conservatively assumes that all roots are within the Contaminated Zone. As has been discussed in Section IV, when all roots lie within the Contaminated Zone, the apparent sensitivity of both the thickness of the contaminated zone, and the depth of roots vanishes.

Depth of Soil Mixing Layer.
As discussed in Section IV on Sensitivity Analysis, the Depth of Soil Mixing Layer has been chosen to be the same as the Thickness of the Contaminated Zone, 0.15 meters, in order to conservatively address the impact of this parameter on the amount of material available for resuspension.

Plant Transfer Factors.
The risk and dose calculations handle plant transfer factors somewhat differently. The risk calculations sum the individual plant ingestion sub-pathways, so that different plant transfer factors can be applied to each plant category (leafy vegetables, non-leafy vegetables and fruits, and grains). RESRAD needs a single value as an input for a soil-to-plant transfer factor.

Dr. Ward Whicker recommends basing root uptake values on results reported in a study at the Savannah River Plant (Whicker, et al, 1999) measured in terms of weight of dry plants per weight of dry soil. The root uptake factor for non-leafy vegetables will be applied to fruits and grains as well.

Root Uptake in Dry Plant Weight per Dry Soil Weight
(derived from Whicker et al, 1999)

PLANT CATEGORY	Pu-239/240	Am-241
Leafy vegetables	2.3E-03	5.3E-02
Non-leafy vegetables (average)	2.5E-04 *	4.5E-03

* The discrepancy with the later value provided by Whicker (Whicker, 2001) of 1.9E-04 is due to a difference in averaging approaches and results in a slightly more conservative value.

Conversion factors listed in Baes, (Baes et al, 1984) can be used to convert these values to wet plant weight per dry soil. Wet plant weight is the form in which food consumption is reported and is the form required as input to the risk equations and the RESRAD code. These dry to wet-weight conversion factors are based on actual measurements of the weight of fresh plant tissue compared to the weight of dried plant tissue. The Baes report listed an overall average value of 0.428, which is weighted based on U.S. production during the 1980's for each plant. This heavily weights the overall average in favor of grains such as wheat, barley and rice, which are not common components of backyard gardens. The working group also recognized that production-based weighting may change with time. Therefore, the working group developed simple average values for each plant category, based on selected plants typically grown in Colorado. An arithmetic average of 17 conversion factors for root vegetables, fruits, corn and peas is 0.16 and the average of conversion factors for 3 grains is 0.89. The reported conversion factor for leafy vegetables is 0.07. Converted uptake values are listed in the following table:

Root Uptake Converted to Wet Plant Weight per Dry Soil Weight

PLANT CATEGORY	Pu-239/240	Am-241
Leafy vegetables	1.6E-04	3.7E-03
Non-leafy vegetables and fruits	4.0E-05	7.2E-04
Grains **	2.2E-04	4.0E-03

** The value for grains applies only to the EPA Risk Assessment Methodology and is not required as a RESRAD input.

To develop radionuclide-specific soil-to-plant transfer factors for RESRAD input, the converted transfer factors have been weighted by the homegrown proportions for each plant category. Based on data from the Exposure Factors Handbook (EPA, 1997), dietary intake from leafy vegetables is approximately 15% and from non-leafy vegetables and fruits is 85%. Because data are not available to distinguish the dietary proportion of grains, grains are not included in the plant transfer factor equations. A working group assumption is that homegrown grains make up only 1 percent of the total grain consumption, so excluding grains will not significantly impact the result.

Radionuclide-Specific Plant Transfer Factors:
Pu-239/240 => (1.6E-04)(.15) + (4.0E-05)(.85) = 5.8E-05
Am-241 => (3.7E-03)(.15) + (7.2E-04)(.85) = 1.2E-03

These values compare with the current RESRAD default of 1.0E-03 for both Pu and Am.

External Gamma Shielding Factor

The External Gamma Shielding Factor is the ratio of the external gamma radiation level indoors on site to the radiation level outdoors on site. It is based on the fact that a building provides shielding against penetration of gamma radiation. The previous Superfund Risk Assessment guidance used a default value of 0.8 for the shielding factor for gamma radiation due to being inside a house. A shielding factor of 0.8 implies that an individual would receive 80% of the gamma dose available to someone outdoors. This value was based on empirical studies of the attenuation of natural background radiation (including terrestrial sources, highly penetrating cosmic rays, and radiations emitted by the building materials themselves). The default value was recently revised to 0.4 in the Soil Screening Guidance for Radionuclides: Technical Background Document (EPA 2000,). The basis for the revision is a review of newer literature, including studies of shielding from fallout and from nuclear power plant releases. This review of additional studies is summarized in the EPA report, "Reassessment of Radium and Thorium Soil Concentrations and Annual Dose Rates" (USEPA, 1996). In addition to the incorporation of additional information, the new default value is lower because it considers only the terrestrial sources of natural background and excludes the cosmic ray and building material sources. This more correctly assesses the shielding afforded from the building from contamination in soil. Based upon this more recent work, the Working Group selected the value of 0.4 for this parameter.

Mass Loading

Mass loading is a sensitive parameter in the RESRAD and EPA standard risk methodology calculations. The exact scenarios being considered, from an air quality perspective, do not exist in previous experience either on the site or elsewhere, and thus historical data cannot be used directly to infer either a deterministic or probabilistic mass loading appropriate to these scenarios. Instead, the RSAL Working Group had to examine other sources of information from which to derive a mass loading estimate.

The Working Group was able to derive a great deal of information from EPA's "Compilation of Air Pollutant Emissions Factors" (AP-42)(EPA, 1995) regarding several sources whose influence might be considered when developing a mass loading distribution for the RSAL calculations. Emission sources or activities that were examined included garden tilling, use of recreational vehicles/horses, and fugitive dust due to passive wind-blown disturbance of soil. The latter influence was examined in detail, including the modifying influences of prairie fire and precipitation. The wind-blown dust that would be an aftermath of a widespread prairie fire was characterized using site-specific wind tunnel measurements.

Once the behaviors of these source influences were characterized, the emission characteristics were integrated into a model that describes the frequency of occurrence and the effect of each source influence on the airborne soil-mass concentrations, i.e. the mass loading.

In the sections that follow we describe the various source influences, the method used to integrate those influences into a frequency distribution describing mass loading, and the mass loading itself.

Mass Loading Influences

Garden Tilling- In the Rural Resident Scenario and in the Wildlife Refuge Worker Scenario there exists a potential for some gardening-type activities. In both cases, the activity would be limited to relatively small areas of the Site. In the Wildlife Refuge Worker Scenario, this activity would not be expected to occur on contaminated soil, but under a case of failed institutional controls, as in the Rural Resident Scenario, gardening could occur on such soils. The rural resident is posed to reside on a relatively small plot of approximately five acres, all contaminated. The Working Group proposed that as much as one acre of that land might be gardened. The area would be prepared for the crop through several tilling cycles and the remains of the crop would be turned under at the end of the growing season. AP-42, Section 11 of the fourth edition (EPA, 1985), provides emission calculations for such activities.

The emission factor for agricultural tilling depends on several individual parameters, the silt content of the soil, the maximum particle size of interest, the tillage acreage and the number of times tilled in the period of interest. For our purposes, the silt content is 50% (Kaiser-Hill, 2000) and the particles of interest are those less than 10 μin diameter, i.e. those that can be readily inhaled during the activity. The tilled acreage is 1 acre with three tilling cycles in a year. The resulting increase in emissions is comparable in magnitude to the typical emissions from wind-blown fugitive dust off the same surface when covered with normal prairie vegetation; in other words, the mass loading is increased no more than a factor of two. Considering that irrigation of the vegetable crop will actually result in fewer emissions than a normally unirrigated surface, the factor of two is considered a reasonable limit on increased emissions over the crop year.

Recreational Vehicle/Horses- The Working Group considered the possibility that horses or light recreational-type utility vehicles might be operated on the site. Such activity could constitute a dust emission source for the RSAL mass-loading calculation. Fugitive dust emissions from horses were not found characterized in the literature, however, dust emissions from treaded vehicles are. If one considers a horse to be similar to a light recreational utility vehicle, or is simply interested in the vehicle emissions, then this calculation applies. Since these activities, or others very similar, could be associated with any of the scenarios being characterized in these RSAL calculations, this assessment is applicable to each of them.

Consider the parameters needed to estimate light utility vehicle emissions; they are the mass of the vehicle, the number of surfaces in contact with the soil, the average speed of the vehicle, and the distance traveled (EPA, 1995, page 13.2.2). As a surrogate, a horse and rider may have a mass of about 400 kg, have four surfaces in contact with the soil (repetitive hoofed contact with the ground is not unlike repeated cleated contact with the ground from a vehicle tread), travel at an average speed of about 5 miles per hour, and exercise for about half an hour per session (not atypical of a utility farm vehicle, itself). If the vehicle (horse) were operated this way twice per week, the expected emissions from such an activity would be approximately 13 kg/year, about 1/3 the emissions from fugitive dust from a 5 acre area in the absence of any soil disturbance. Even with daily activity, the emissions would be comparable.

Considering the combined effects of gardening and recreational vehicle/horseback riding, the average mass-loading in the area around the activities might be expected to increase by as much as a factor of two compared to the fugitive emissions that would be present without such activities. The Working Group took this factor into account when building the mass-loading distribution, assuming that such activities would occur with the same probability in any single year.

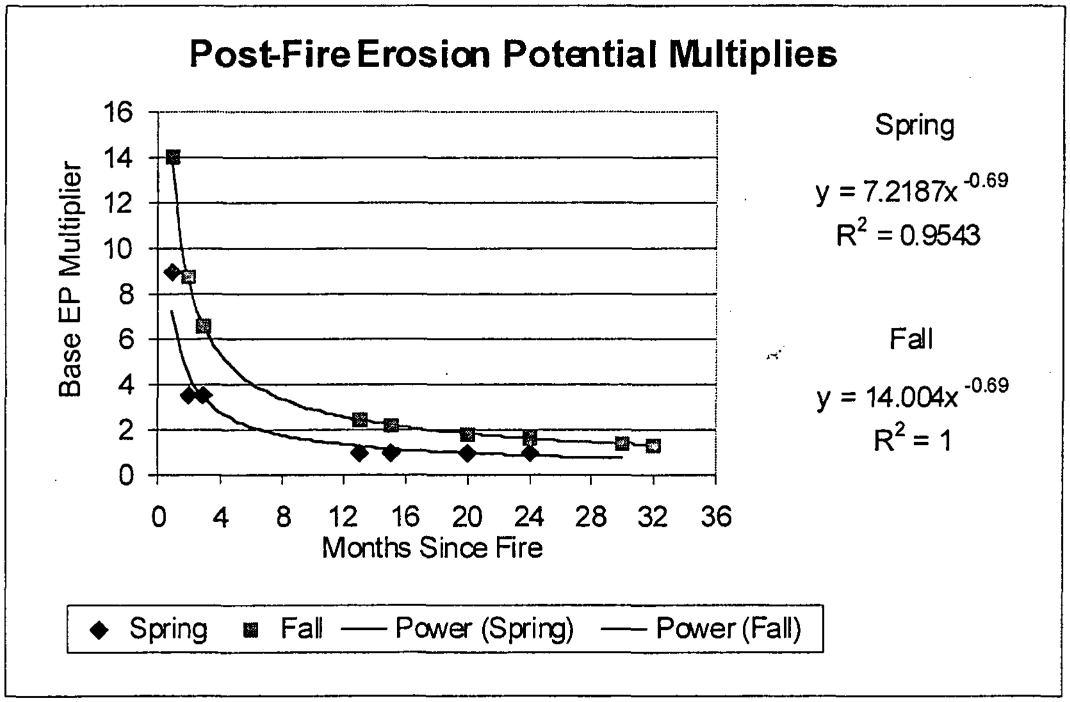
Fugitive dust under normal conditions at Rocky Flats- Rocky Flats experiences nearly continuous winds, varying in speed from near calm (infrequently) to more than 40 m/s on some occasions in the late winter and early spring. The median annual average wind speed at the Site is about 4.2 meters per second, based on more that 25 years of site-specific meteorological data. One of the predictable influences of these sustained winds is a relatively large contribution to mass loading from wind-blown soil erosion. Related to this is the observation that the majority of radionuclide emissions from the site come from the resuspension of contamination attached to soil particles, mostly from the eastern lip of the Industrial Area and the eastern and south-eastern Buffer Zone of the Site. Very little of the observed emissions originate from the building stacks.

Effect of prairie fire on contaminant resuspension- Concern was raised during the 1996 RSAL peer review performed by RAC Corporation that a prairie fire at the Site could have considerable influence on the amount of soil eroded into the air following such a fire. As a result of this concern, and the recognition that no data could be found in the literature that characterize the post-fire effects of a prairie fire, the Site engaged Midwest Research Institute (MRI) to perform wind-tunnel-based soil erosion measurements. The measurements were performed on burned vegetated surfaces following a controlled burn conducted at the Site in CY2000. The erosion potential was measured at several intervals over the months immediately following the controlled burn to develop a profile that characterizes the rate of recovery of the burned area. It was postulated that the burned area would have a much higher erosion potential in the first few days or weeks following the fire, but would exhibit continuously improving erosion inhibition as the vegetation grew back over the burned, denuded soil.

The results of the wind-tunnel measurements confirmed that the erosion potential would decrease rather quickly with time following the controlled burn. Effects of soil moisture on erosion potential were also evident in the same set of measurements. The wind-tunnel work has been described in detail in two final test reports from MRI (MRI 2001a and MRI 2001b). The analysis of these data is described below.

The MRI controlled burn report (MRI 2001a) provides three sets of post-fire measurements to demonstrate the effects of vegetative recovery on the erosion potential of the surface soils. When these erosion curves are compared, they suggest the wind-blown erosion is reduced to less than one-third of its maximum within three or four months of the fire. If this behavior is fitted to a simple power curve, shown as Figure A-1, the results show that the burned area will recover its dust mitigation characteristics completely within six to twelve months following the fire, except for the possible mitigating effects of thatch which will not be present within such a short period. (The presence of thatch would be more important in areas denuded of growing vegetation as might occur during a drought, and would not tend to be an important factor in overgrown areas.)

Figure A-1 Mathematically fitted erosion-potential recovery curves following spring or fall prairie fires at Rocky Flats



Had this same fire occurred in the fall or early winter, the recovery period would have been lengthened. The resulting mass-loading multiplication factor associated with these late-season fires is 4.74, as derived from the fall curve shown in Figure A-1. This factor was estimated using the same arguments as with the spring fire but interpolated over a period of 24 months, to account for the arrested period of growth during the winter months immediately following the late-season fire. The same precipitation adjustments were applied to each month for the first year of recovery, and the average emission factor was calculated. The initial emissions from a late-season fire will be somewhat higher than for the spring fire, evidenced by the wind tunnel recovery curve for the June measurements (taken during a relatively dry period, representative of soil conditions in Fall).

Details of how these curves were used to derive the empirical mass-loading multipliers can be seen in TableA-1, below. In order to calculate an annual average increase attributable to a prairie fire, each month's emission potential (from the fitted curve) is then adjusted by a factor that accounts for the expected precipitation for that month and the average emission potential for all periods are averaged. The average increase in emissions associated with this rapid recovery is approximately 2.5 times the emissions associated with similar adjacent areas of unburned grasslands used as a control on the measurements, as indicated in Table A-1. The factor actually used in the mass-loading calculations is 2.51.

TABLE A-1 Calculation of Mass Loading Multiplier, highlighted numbers are results for spring and fall burns, respectively.

Time months	Spring Monthly Contribution	Fall Monthly Contribution	Annual Precipitation Factor	Spring Monthly Contribution. w/precipitation	Fall Monthly Contribution w/precipitation
1	0.75	1.17	0.926	0.69	1.08
2	0.29	0.72	0.926	0.27	0.67
3	0.29	0.55	0.926	0.27	0.51
4	0.23	0.45	0.926	0.21	0.42
5	0.20	0.38	0.926	0.18	0.36
6	0.17	0.34	0.926	0.16	0.31
7	0.16	0.30	0.926	0.15	0.28
8	0.14	0.28	0.926	0.13	0.26
9	0.13	0.26	0.926	0.12	0.24
10	0.12	0.24	0.926	0.11	0.22
11	0.12	0.22	0.926	0.11	0.21
12	0.11	0.21	0.926	0.10	0.19
	2.72	5.12		2.51	4.74

Effects of precipitation- In the preceding section, the effects of precipitation on erosion potential for airborne fugitive dust emissions were described briefly, concerning in particular the mediating effects of snow cover. AP-42 describes similar effects for rainfall precipitation. As a means of estimating fugitive emissions, days with rain exceeding 0.01 inches are treated as though their emissions are zero. As we have described previously, days with snow cover can be treated the same. The question might be raised then – what is the effect on fugitive dust during periods of drought? (Periods of excessive rainfall were also examined, but their influence is not considered as important to the discussion as periods of deficient rainfall.)

Literature from The National Drought Mitigation Center, headquartered at University of Nebraska - Lincoln, (NDMC,1995), suggests that the onset of drought is marked by a sustained period with rainfall at levels 75% or less compared to that normally experienced. This is preferably based on a 30-year or greater meteorological history. At Rocky Flats, a 37-year meteorological history has been reviewed and summarized (EG&G, 1995)) and provides a good basis for assessing the potential effects and frequency of occurrence of drought-like conditions. In addition, data from state publications and databases (CSU, 2000) provide insight into the occurrence of drought in the State, as a whole. From site-specific meteorological data, we were able to infer that ROCKY FLATS could experience drought-like conditions about 20 % of the time. During those periods, there are roughly 40% fewer days with rainfall that may exceed 0.01 inches, compared to a median estimate of 78 days with such amounts. This suggests that the dry conditions might be characterized by emissions that are increased by about 11% based on this calculation that inhibits emissions on days with greater than 0.01 inches of rain. The number used to characterize this condition in the mass loading calculation was 14%, based on a linear fit to the precipitation data with one biased month removed. (The month of May, with its extreme precipitation, does not appear to be representative of the typical behavior for this parameterization.) It is worth noting, that the emissions would be expected to increase by about 27% should there be no rainfall, and no other contribution to increased emissions. Zero rainfall was not considered a feasible condition to assess.

To summarize, the drought-like conditions that might be observed to increase emissions at Rocky Flats would occur about 20% of the time and would result in emissions increased by about 11 % or more. Because of the uncertainty in this estimate due to one apparently non-representative month, the emissions were considered to increase by 14%.

Building a Mass-Loading Distribution

The information described above was combined with site-specific and Statewide PM-10 data to build mass loading distributions for both PM-10 and TSP air mass concentrations.

Site-specific PM-10 and TSP mass concentrations

Appendix F provides the site-specific PM-10 data obtained from the CDPHE five-station network. The data are described by a minimum concentration of 9.4 $\mu g/m^3$, a maximum concentration of 16.6 $\mu g/m^3$ and a median concentration of 11.6 $\mu g/m^3$ Data from the site's RAAMP network have been used to relate the PM-10 data to TSP data, specifically the relative distribution of plutonium between PM-10 and TSP. Data collected since 1994 show a relatively consistent trend with the larger TSP fraction having about 2.5 times the activity of the airborne material smaller than 10 μm aerodynamic diameter.

State-wide PM-10 mass concentrations

Appendix F also provides a six-year set of PM-10 mass concentrations from throughout Colorado. These data are representative of air quality in areas most likely to be impacted by industrial, agricultural and urban emissions. They could be considered as a probable representation of the likely extremes of air quality that might be observed at Rocky Flats in the future, should the area be developed residentially or commercially. These PM-10 mass concentrations are described by a distribution whose minimum is 6.7 $\mu g/m^3$, maximum is 51.4 $\mu g/m^3$, and median concentration is 20.3 $\mu g/m^3$.

Building a frequency distribution

Lacking a set of data that can serve as an adequate surrogate for all of the possible conditions that might exist in future scenarios being modeled for Rocky Flats, it is possible to develop a descriptive statistical model of mass concentrations. To build this frequency distribution, it is first necessary to describe the events that will provide the significant influences on the mass concentrations, including their frequency of occurrence. These have been described physically in the last section.

In order to build a distribution of mass loading, a starting value must be chosen. For these calculations, the median state PM-10 value of 20.3 $\mu g/m^3$ was chosen because it seems to be a representative value for conditions that might be experienced in the future at Rocky Flats. To further validate this assumption, we considered what might happen to the median site-specific value, 11.6 $\mu g/m^3$ if it were increased by the factors that might be applicable for gardening or recreational horseback riding, as described earlier. The median value would be increased by about a factor of two under these several conditions, confirming the choice of the statewide median as a reasonable starting point.

Describing them again here, related to some frequency of occurrence, we present the following model. Normal conditions, without significant drought and wildfire effects prevail. With some regular frequency, these normal conditions are modified by the occurrence of periods with deficient rainfall, causing an increase in airborne dust. In addition these normal events may be influenced by occasional wildfire events. For the purpose of developing the model, the periods with deficient rainfall were assumed to occur about 25% of the time, with an increase in air concentration of about 14%. Fire events were assumed to occur about 10% of the time, with increases in air concentrations of between 151% and 374%, divided equally between spring events (representing fast recovery periods) and fall events (representing slow recovery periods).

Regarding conditions that might mitigate some of these effects, it might be argued that a wild-fire would not occur in an area that contained a cultivated garden. The Working Group could not eliminate such an event, considering that the wildfire might consume the vegetation adjacent to the garden plot, but not burn the plot itself, due to irrigation. Likewise, the presence of a cultivated garden would not effectively mitigate the dust-laden effects of a period of low rainfall. The environmental conditions that characterize the resulting mass loading are summarized in the following table.

Table A-2 Frequency and weighting associated with each annual environmental condition.

Fire	Frequency	Weighting
No fire, normal precipitation	0.75	1
No fire, dry conditions	0.25	1.14
Spring fire, normal precipitation	0.75 x 0.05 = 0.0375	2.51
Spring fire, dry conditions	0.25 x 0.05 = 0.0125	2.87
Fall fire, normal precipitation	0.0375	4.74
Fall fire, dry conditions	0.0125	5.42

Calculated Distribution - Mass Loading for Inhalation

Table A-3 shown below, summarizes the calculations that result from combining these weightings with the median PM-10 mass concentration derived from the statewide air quality data contained in the AIRS database.

Table A-3 Mass Loading derivation, tabulated

Fire	Weight	Frequency	Precipitation	Weight	Frequency	Grand Weight	Grand Frequency	Mass Loading	Cumulative Frequency
No fire	1	0.9	Normal Precipitation	1	0.75	1	0.6750	20.2	0.338
No fire	1	0.9	Dry Conditions	1.14	0.25	1.14	0.2250	23.1	0.788
Spring fire	2.51	0.05	Normal	1	0.75	2.51	0.0375	50.7	0.919
Spring fire	2.51	0.05	Dry	1.14	0.25	2.87	0.0125	58.0	0.944
Fall fire	4.74	0.05	Normal	1	0.75	4.74	0.0375	95.7	0.969
Fall fire	4.74	0.05	Dry	1.14	0.25	5.42	0.0125	109.5	0.994

These six mass loading values provide a set of input values for the "continuous linear" distribution input capability of RESRAD. RESRAD requires that 0^{th} and 100^{th} percentile values be input along with these intermediately distributed values. The 0^{th} percentile mass loading was chosen to be 9.4 $\mu g/m^3$, consistent with the lowest annual average PM-10 value observed in the samplers around the Site. The 100^{th} percentile mass loading was chosen based on the highest value observed in the statewide data, increased by a factor of about 4, midway between the values that would be obtained from spring or fall fire scenarios; 200 $\mu g/m^3$ was chosen. The same input values were used for the EPA STANDARD RISK METHODOLOGY calculations after passing them through a fitting routine to generate an equivalent mathematically formulated distribution.

Mass Loading for Foliar Deposition

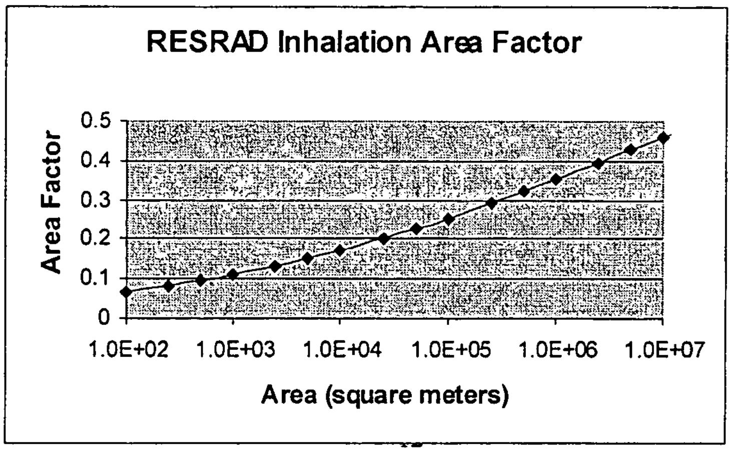
In addition to the mass loading for inhalation, the mass loading associated with deposition of contaminated dust onto garden fruits and vegetables must also be calculated. As noted earlier, the radioactivity of total suspended particulate matter is about 2.5 times the radioactivity of the finer less-than-10 μm fraction. The mass loading for foliar deposition can be simply derived by multiplying each mass concentration given in Table A-3 by this constant factor. The 0^{th} and 100^{th} percentile values are calculated the same way.

Differences Between EPA STANDARD RISK METHODOLOGY and RESRAD Regarding Calculation of Contaminated Fraction of Inhaled Particulate Matter (Contaminated Mass Loading)

RESRAD 6.0 uses the mass loading parameter as input to its inhalation dose and risk calculations. This input is multiplied by a quantity called the "Area Factor", that takes into account the amount of particulate matter in the air that may be contaminated by wind-eroded contaminated soil from the area of contamination being considered in the modeling calculations. The area factor is sensitive to both the area of contamination and the wind speed, increasing in magnitude with increasing area, and decreasing with increasing wind speed. Figure A-2 shows the behavior of the Area Factor as a function of contaminated area, for a 5 m/s wind speed, similar to the annual average wind speed for Rocky Flats.

EPA STANDARD RISK METHODOLOGY uses a constant mass loading in its calculations of inhalation risk, assuming all of the airborne particulate matter is contaminated. If the RESRAD and EPA STANDARD RISK METHODOLOGY calculations of contaminated mass loading are compared, the RESRAD input will be reduced relative to the EPA STANDARD RISK METHODOLOGY input by the Area Factor multiplier. In other words, for the 300 acre area considered in the scenarios being reported in this document, the contaminated mass loading is about 37% of the contaminated mass loading used in EPA STANDARD RISK METHODOLOGY.

Figure A-2 Area Factor used to calculate the contaminated mass loading due to wind-eroded soil.



Soil Ingestion Rate in Children (ages 1 - 7 years)
A review of the literature on soil ingestion rates was conducted in order to develop a probability distribution function (PDF) for use in Monte Carlo simulations. The PDF is intended to characterize interindividual variability in long-term average soil ingestion rates among children. The following discussion explains the general fecal tracer study methodology used to indirectly assess ingestion rates. The most relevant empirical data are summarized, and justification for the most applicable distribution for Rocky Flats is offered. While the goal is to characterize interindividual variability in ingestion rates over long time periods (e.g., years), the study designs capture short periods (e.g., days), which require simplifying assumptions to extrapolate beyond the observed results. Uncertainties associated with these assumptions are outlined.

The published literature describing statistical analyses of childhood soil ingestion rates is sizable and often very technical. To facilitate an understanding of how a PDF was developed for use in the risk assessments presented in this document, a separate reference list and glossary of technical terms is included at the end of this section.

Probability Distribution
The following probability distribution was developed for use in probabilistic risk and RSAL calculations:

IRs_child ~ Truncated Lognormal (47.5, 112, 0, 1000) mg/day

The truncated lognormal distribution is defined by four parameters:

- arithmetic mean 47.5 mg/day
- standard deviation 112 mg/day
- minimum 0 mg/day
- maximum 1000 mg/day

Figure A-3. Probability density function (PDF) and cumulative distribution function (CDF) views of the probability distribution for child soil ingestion rate (mg/day).

Summary Statistics	**IRsd (mg/day)**
mean	47.5
stdev	112
50th %ile	18.5
75th %ile	46.8
90th %ile	107.5
95th %ile	177.0
95.8th %ile	200.0
99th %ile	450.7
Max	1000

Uncertainties in the Probability Distribution

The methodology and data analysis associated with the published estimates of child soil ingestion rates is complex. An overview of the methodology is given below in order to highlight the major assumptions and uncertainties associated with the development of the distribution.

Fecal Tracer Methodology for Estimating Soil Ingestion Rate

Empirical estimates of soil ingestion rates (IR_{soil}) in children have been made by backcalculating the mass of soil and/or dust a subject would need to ingest to achieve a tracer element mass measured in collected excreta (i.e., feces and urine) (Calabrese et al., 1996). The general expression for the trace element ("tracer") mass balance is given by Equation 1:

$$[tracer]_{out} - [tracer]_{in,nonsoil} = [tracer]_{in,soil}$$

where $[tracer]_{out}$ is the average daily tracer mass (_g) measured in feces and urine, $[tracer]_{in,\,non\text{-}soil}$ is the average daily tracer mass measured in non-soil ingesta (i.e., food, water, toothpaste, and medicines), and $[tracer]_{in,\,soil}$ is the estimated average daily tracer mass in ingested soil. Dividing all terms by the measured tracer concentration in soil (_g/g) yields an estimate of the average daily soil ingestion rate, as given by Equation 2:

$$\frac{[tracer]_{out} - [tracer]_{in,nonsoil}}{\dfrac{[tracer]_{soil}}{[soil]}} = \frac{[tracer]_{in,soil}}{\dfrac{[tracer]_{soil}}{[soil]}} = [soil] = IR_{soil}$$

Empirical Data

Three seminal studies, briefly summarized below, used this mass-balance approach and were considered appropriate for quantifying variability and uncertainty in IR_{soil}. Pathways for non-soil/non-food intake of tracers (e.g., inhalation and dermal absorption) and excretion (e.g., sweat and hair) were not measured in these studies and are thought to be minor components of the overall tracer mass balance (Barnes, 1990).

(i) **Calabrese et al. (1989)** - Eight trace elements (Al, Ba, Mn, Si, Ti, V, Y, and Zr) were measured in a mass-balance study of 64 children ages 1 to 4 years over 8 days (i.e., 4 days per week for 2 weeks) during late September and early October. Participants represent a nonrandom study population selected from day-care centers and volunteer families in an academic community in Amherst, MA. A single composite soil sample was collected from up to 3 outdoor play areas identified by parents as locations where subjects spent the most time. Similarly, indoor dust samples were vacuumed from floor surfaces that parents reported to be common play areas during the study. Each week, duplicate food samples were collected for 3 consecutive days, and fecal samples (excluding diaper wipes and toilet paper) were collected for 4 consecutive days for each subject. A total of 128 subject-week estimates of IR_{soil} were made. Also, since food and fecal samples were collected on multiple days per subject, a total of 439 subject-day estimates of IR_{soil} were also made (Stanek and Calabrese, 1995b). For each subject-week-day, a maximum of 8 estimates of IR_{soil} were made, each estimate corresponding to a unique trace element.

(ii) **Davis et al. (1990)** - Three trace elements (Al, Si, and Ti) were measured in a mass-balance study of 101 children ages 2 to 7 years over 4 consecutive days during the summer. Participants represent a random sample of the population in a three-city area of southeastern Washington State. A single composite soil sample was collected from outdoor play areas identified by parents. Indoor dust samples were collected by vacuuming floor surfaces of the child's bedroom, the living room, and the kitchen, as well as by sampling the household vacuum cleaner. Information on dietary habits and demographics was collected in an attempt to identify behavioral and demographic characteristics that influence soil ingestion. Although duplicate food and fecal samples (including diaper wipes and toilet paper) were collected on a daily basis, samples for each individual were pooled to derive a one-week average estimate of IR_{soil}. A total of 101 subject-week estimates of IR_{soil} were made. For each subject-week, a maximum of 3 estimates of IR_{soil} were made, each estimate corresponding to a unique trace element.

(iii) **Calabrese et al. (1997a)** - Eight trace elements (Al, Si, Ti, Ce, Nd, La, Y, and Zr) were measured in a mass-balance study of 64 children ages 1 to 3 years over 7 consecutive days during September. Participants were selected from a stratified simple random sample of approximately 200 households from 6 geographic areas in and around Anaconda, MT. A single composite soil sample was collected from up to 3 outdoor play areas identified by parents as locations where subjects spent the most time. Similarly, indoor dust samples were vacuumed from floor surfaces that parents reported to be common play areas during the study. Duplicate food and fecal tracer element samples were collected for 448 and 339 subject-days, respectively. A total of 64 subject-week estimates of IR_{soil} were made; subject-day estimates of IR_{soil} have recently been published (Stanek and Calabrese, 1999; 2000; Stanek et al., 2001a). Three trace elements (Ce, La, and Nd) were not used to estimate IR_{soil} because soil concentrations of these

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elements were found to vary by particle size (Calabrese et al., 1996). For each subject-week, a maximum of 5 estimates of IR_{soil} were made, each estimate corresponding to a unique trace element. Final soil ingestion estimates are based on soil particle size < 250 _m (as opposed to 2 mm).

INTERPRETATION OF INTER-TRACER VARIABILITY IN SOIL INGESTION

Trace elements were selected for estimating soil ingestion in these mass-balance studies because they are natural constituents of soil, present in relatively low concentrations in food, poorly absorbed in the gastrointestinal tract, and not inhaled in appreciable amounts (Barnes, 1990). Theoretically, each trace element should yield the same estimate of daily soil ingestion using Equation 2. However, the following sources of measurement error are attributed to the high inter-tracer variability and low precision of recovery observed for many subject-days in each study:

- High element concentration in food, yielding a high food-to-soil (F/S) ratio (Calabrese and Stanek, 1991);
- Variability in food transit times between subjects and between subject-days for a given child resulting in input/output misalignment errors, and lower precision of recovery for elements with higher F/S ratios (Stanek and Calabrese, 1995a); and
- Incomplete collection of both inputs (e.g., additional non-soil sources of tracer) and outputs (e.g., fecal samples on diaper wipes and toilet paper; urine samples for elements with low fecal-to-urine ratios).

The adult validation study by Calabrese et al. (1989; 1990) demonstrated that negative soil ingestion estimates occur more frequently for trace elements with high F/S ratios. At a low dose of soil (100 mg/day), 7 of 48 (15%) subject-days displayed negative IR, while at a high soil dose (500 mg/day), no subjects displayed negative IR. The adult study by Calabrese et al. (1997a), which used a slightly different set of trace elements, demonstrated a sufficiently high recovery for most elements to quantify ingestion rates in the range 20 to 500 mg/day. These results may also apply to children, keeping in mind potential differences in the following areas among different age groups: gastrointestinal (GI) transit times, absorption efficiencies, F/S ratio, and variability in daily tracer ingestion (Calabrese and Stanek, 1991). For the studies with children, negative IR estimates were observed on 12 to 44% of subject-days (depending on the trace element) by Calabrese et al. (1989); 12 to 32% by Davis et al. (1990); and approximately 55% (preliminary assessment of Al and Si) by Calabrese et al. (1997a). Given that high inter-tracer variability in subject-day estimates of IR_{soil} is a function of both tracer-specific properties and input/output errors, it is unlikely that a reliable estimate of IR_{soil} for all subject-days can be derived from any single trace element. This is confirmed by the differences in estimates of ingestion rates among different tracers. For example, tracer-specific estimates of median IR_{soil} in the Calabrese et al. (1989) study range by an order of magnitude (i.e., 9-96 mg/day). The following two methodologies have been developed to identify the set of trace elements that is likely to provide the most reliable estimate of IR_{soil}.

(A) **Best Tracer Method (BTM)** - Each subject-week estimate of IR_{soil} is based on the trace element(s) with the best (i.e., lowest) F/S ratios for that week (Stanek and Calabrese, 1995a). This approach reduces the effect of transit time errors (i.e., poor temporal correspondence

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between food and fecal samples). Potential bias from other sources of error for specific tracers may be reduced by estimating the median of multiple tracers with low F/S ratios for a subject-week. Stanek and Calabrese (1995a) recommend estimating the distribution of IR_{soil} based on the median of the 4 best tracers for each subject-week. Using this approach, data from the Calabrese et al. (1989) and Davis et al. (1990) studies were combined to yield 229 subject-week estimates of IR_{soil} representing 165 children between the ages of 1 and 7.

(B) **Daily Estimate Method**- A single estimate of IR_{soil} is made for each tracer-subject-day for each child (Stanek and Calabrese, 1995b; 2000). A maximum of 8 such estimates (one per tracer) was determined for each of 64 children in the Calabrese et al. (1989) study. This approach establishes a set of criteria to identify tracer-subject-day estimates that may be unreliable for each subject-week, based on the relative standard deviation (RSD) given by Equation 3:

$$\Delta_i = \max(50,\ d_i e^{[1.5-0.35\ln(d_i)]})$$

$$\delta_i = \left| d_{ij} - d_i \right|$$

$$RSD_i = \frac{\Delta_i}{\delta_i}$$

where d_i is the median IR_{soil} for the i^{th} day of a given subject-week, d_{ij} is the IR_{soil} for the j^{th} tracer on the i^{th} day of a given subject-week, Δ_i is the maximum of either 50 mg/day or a function of d_i, and δ_i is the absolute value of the difference between a single tracer element and the median among the group of tracers on a given day. Stanek and Calabrese (1995b) limited the maximum value of _i to 50 mg/day to reduce any bias associated with low median estimates of IR_{soil}. If, for a given d_i, _i > _i, then RSD < 1.0 and element j is identified as an outlier estimate of IR_{soil}. The median of the remaining tracers for each subject-day was considered the best estimate of IR_{soil}.

The Daily Estimate Method attempts to correct for positive and negative mass-balance errors at the level of the subject-day. This approach reduces the effect of transit time errors by directly linking the passage of food and fecal samples for each daily estimate. Like the BTM approach, it reduces tracer-specific source errors by calculating the median of multiple tracer estimates. An advantage of this approach over BTM is that it also allows for an estimate of intraindividual (within subject) variability in IR_{soil}. After applying the RSD exclusion criteria to the Calabrese et al. (1989) Amherst data, daily estimates of IR_{soil} (based on the median of tracer-specific estimates) were available for at least 4 days for all subjects, and at least 6 days for 94% of the subjects (Stanek and Calabrese, 1995b). Assuming each subject's daily IR_{soil} is log normally distributed, subject-specific parameters for lognormal PDFs were defined based on the mean and variance of the 4 to 8 daily IR_{soil} values. Each lognormal PDF was then used to define daily ingestion rates over a 365-day period. The use of a lognormal distribution (instead of other right-skewed distribution) is an acknowledged source of uncertainty that was not explored further due to the limited number of days of data for each individual (Stanek and Calabrese, 1995b). A similar approach could not be applied to the Davis et al. (1990) data because daily estimates of IR_{soil} were combined to define subject-weeks. This approach was also applied to the Calabrese et al. (1997a) Anaconda data (Stanek and Calabrese, 2000)as summarized in Table 1 below.

EVALUATION OF SHORT-TERM AND LONG-TERM EDF'S FOR SOIL INGESTION RATE

As of 1994, estimates of childhood soil ingestion rates from short-term studies were assumed to be representative of long-term rates. U.S. EPA (1994a,b) recommended a default central tendency estimate of IR_{soil} = 135 mg/day for ages 12 to < 48 months based on a review of mean tracer-specific estimates given by Binder, Sokal, and Maughan (1986), Clausing, Brunekreef, and Van Wijnen (1987), Calabrese et al. (1989), and Davis et al. (1990). Currently, only two of the mass balance fecal tracer studies are suitable to estimate daily soil ingestion rates needed to develop estimates of long-term average rates: 1) Amherst, MA (Calabrese et al., 1989; Stanek and Calabrese, 1995b) and 2) Anaconda, MA (Calabrese et al., 1997a; Stanek and Calabrese, 2000; Stanek et al., 2001a). Table 1 summarizes the estimates of interindividual variability in IR_{soil} derived from the results of the three soil ingestion studies with children that used a mass-balance approach. An empirical cumulative distribution function (EDF) was developed from the summary statistics derived by the Daily Estimate Method (i.e., Daily Mean, 1+) applied to both the Amherst and Anaconda data. These studies and the statistical approach were selected for the following reasons:

- The ingestion rates estimated by Calabrese et al. (1989) generally have less uncertainty related to input/output misalignment error than the estimates by Davis et al. (1990). For example, nearly 90% of the subject-weeks reported by Calabrese et al. (1989) had a least 2 trace elements with F/S ratios lower than the lowest F/S ratios reported in the Davis et al. (1990) study (Stanek and Calabrese, 1995a). In addition, although titanium (Ti) has relatively low F/S ratios in both studies, it displayed exceptionally high source error (Calabrese and Stanek, 1995; Stanek et al., 2001a). Consequently, Ti, 1 of only 3 tracers used in Davis et al. (1990), may provide unreliable estimates of IR_{soil}.

- The Daily Estimate Method is preferred over BTM because (1) it identifies sources of potential measurement error at the level of the subject-day rather than the subject-week, and (2) intraindividual variability in IR_{soil} can be quantified and extrapolated over longer time periods. Both of the studies by Calabrese (1989; 1997a) data are amenable to this method, whereas the Davis et al. (1990) estimate of IR_{soil} is for subject-weeks.

Three key assumptions were made in developing a probability distribution from each of the Calabrese data sets using the Daily Estimate Method:

(i) Subject-day estimates of IR_{soil} are reasonable approximations of the combined ingestion of outdoor soil and indoor dust. For simplicity, Stanek and Calabrese (1995b) based all soil ingestion estimates on trace element concentrations in soil, not dust. Theoretically, if concentrations in soil and dust were the same, this approach would correctly account for ingestion from both sources. Relative differences in average concentrations between outdoor soil and indoor dust for the Calabrese et al. (1989) study range from 6 to 55% for different trace elements (Stanek and Calabrese, 1992). Calabrese et al. (1989) proposed apportioning residual fecal tracers using a time-weighting approach, which assumes that soil ingestion is proportional to time spent in a particular location. This is also a

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simplistic approach since soil and dust exposure may vary due to differences in hand-to-mouth activity, weather, and degree of adult supervision. For the data used to generate a PDF for Rocky Flats, no attempt was made to account for potential differences between soil and dust ingestion rates.

(ii) A reasonable upper bound for variability in the long-term average ingestion rate is 1000 mg/day. This assumption reflects an understanding of both intraindividual and interindividual ingestion rates. There is considerable intraindividual variability over a one year period with respect to the frequency and magnitude of soil ingestion. While most children ingest relatively small amounts of soil on most days, occasionally they will ingest large quantities (i.e., > 1000 mg/day). Therefore, while the annual average IR_{soil} may be low for a given child, day-to-day variability may result in several subject-days of high IR_{soil} per year. This hypothesis is suggested by U.S. EPA (1994a) and supported by soil ingestion studies by Calabrese et al. (1989) and Wong (1988), as summarized by Calabrese and Stanek (1993). In the Calabrese et al. (1989) study, one child ingested an estimated 20 to 25 grams of soil on 2 of 8 days (Calabrese, Stanek, and Gilbert, 1993). A second child displayed more consistent but less striking soil pica in which high soil ingestion (1 to 3 g/day) was observed on 4 of 7 days (Calabrese et al., 1997b). Wong observed soil pica (i.e., >1.0 g/day) in 9 of 84 individual subject-days (10.5%) for Jamaican children ages 0.3 to 7.5 years, and at least 1 of 4 days for 5 of 24 (20.8%) children of normal mental capability. One mentally retarded child displayed consistently extreme soil pica over the 4 days (48.3, 60.7, 51.4, 3.8 g soil).

Stanek and Calabrese (1995b) fit individual subject-day estimates from Calabrese et al. (1989) to lognormal distributions to estimate the number of days per year each child might be expected to ingest > 1.0 g/day. Model-based predictions suggest the majority (62%) of children will ingest >1.0 g soil on 1 or 2 days/year, while 42% and 33% of children were estimated to ingest >5 and >10 g soil on 1 or 2 days/year, respectively.

(iii) The developmental period during which the frequency and magnitude of soil ingestion is likely to be the greatest coincides with the period of peak hand-to-mouth activity (i.e., ages 1 to 4 years). It should be noted that empirical data from the mass-balance studies do not provide any evidence that children ages 1 to 4 years ingest more soil than other age groups (Calabrese et al., 1994).

For simplicity, it is assumed that random values selected from this distribution are independent for each time step of exposure. In other words, the latent distribution of individual ingestion rates is assumed to be equal for all individuals in the population. It is more plausible that patterns of soil ingestion rate for an individual are a combination of a latent distribution and some measure of day-to-day variability. Several approaches may be used to simulate this type of exposure pattern in a population. Stanek (1996) combined a latent distribution and response error distribution (for tracers Al, Si, Y) to define an empirical distribution, and then extrapolated the empirical distribution over 365 days. The same approach was employed for the Anaconda data (Stanek and Calabrese, 2000), resulting in 75% lower values for the 365-day average than for the daily values. The resulting distributions are given in Table 1. The response error variance was calculated as the variance in subject-day estimates of $\ln(IR_{soil})$ divided by the

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number of subject-day estimates for a given child. The average response error variance among all 64 Amherst subjects was 0.47, while the average number of subject-days per child was 6.1; therefore, the average standard deviation in daily soil ingestion was approximately 66 mg/day (i.e., SD = exp((0.47^0.5)*6.1).

A similar approach was used to determine variance estimates for the Anaconda data (see Table IV of Stanek and Calabrese, 2000). For purposes of comparison, day-to-day variance in soil ingestion from the Anaconda study (excluding titanium and Tukey far-out) was reported as 9,094 (standard deviation = 95 mg/day), whereas day-to-day variance from the Amherst study (including aluminum, silicon, yttrium, zirconium) was 15,528 (standard deviation = 124 mg/day). These expressions provide the only quantitative measure of intraindividual variability in IR_{soil}.

Extrapolating the empirical distribution over 365 days assumes that the response error variance measured over a short-term period (i.e., subject-week) is the same as the variance over a long-term period (i.e., 365 days). In addition, it assumes that the variance is independent of the average daily IR_{soil} for a given subject week. The upper tail of the empirical distribution may be underestimated if a positive correlation exists between the mean and variance IR_{soil} for a given subject-week. This source of uncertainty could be explored for both Amherst and Anaconda subject-day estimates, but was not for this analysis.

Final Selection of Probability Distribution for Soil Ingestion Rate

The Anaconda data are generally considered to be more representative of the potentially exposed population of children at the Rocky Flats:

- study population is from the West (Montana);
- soil was sieved at 250 _m, a more representative size fraction for particle adherence to hands, and also the size fraction with the least uncertainty in trace element concentrations;
- exclusion criteria for daily tracer estimates resulted in much larger data base of subject-day estimates from which to develop statistical summaries. Exclusion criteria applied to the Anaconda data eliminated estimates based on Ti, and Tukey outlier criteria excluded 18 of 2,984 element-subject days (i.e., 0.45%) compared with 31.9% that would have been eliminated if the Amherst outlier criteria had been applied (Stanek and Calabrese, 2000). Outlier criteria applied to the Amherst study resulted in exclusion of 37.5% of the data (Stanek and Calabrese, 2000).

It is unclear what factors are responsible for study-to-study differences in soil ingestion rates, as was observed between the Amherst and Anaconda cohorts. The empirical distribution function (EDF) is a convenient distribution for characterizing the data sets given a relatively high portion of non-negative values reported for ingestion rate. Non-negative continuous distributions fit to the EDF, such as lognormal, gamma, and Weibull, generally yield poor fits, as discussed by Schulz (2001). Alternatively, a series of mixed distributions or conditional distributions could be developed to make use of parametric distributions such as the lognormal for all non-negative values; these approaches are not presented in the literature.

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While the percentile data can be entered into a Monte Carlo analysis as an EDF, a decision would still be needed regarding the minimum and maximum values of the distribution. Since negative values cannot be employed in a risk assessment, a lower truncation limit of 0 mg/day must be used, and could be assumed to define the minimum. This truncation limit is extended to all of the percentile values corresponding to non-negative ingestion rates. For the Anaconda data, negative values were obtained for the 25th percentile (IR_{soil} = -3 mg/day), which carries through to the best liner unbiased predictor (BLUP) estimates as high as the 7th percentile (see Table A-4) (Stanek et al., 2001a, Table 3). The EDF developed by Stanek et al. for the long-term average ingestion rates was employed in this analysis (last column in Table 1), and can be approximated by a lognormal distribution. For purposes of maximum likelihood estimates of the mean and standard deviation of the lognormal distribution, a maximum of 150 mg/day was applied (slightly greater than the 99th percentile value of 137 mg/day). The choice of the maximum value for truncation can be an important source of uncertainty in risk estimates if there is a high positive correlation between risk and IR_{soil}, especially at the upper tail of the risk distribution (e.g., > 90th percentiles). The goodness-of-fit techniques are also sensitive to the choice of maximum values on the EDF.

Table A-4 Distribution of soil ingestion rates based on different methods of analyzing trace element-specific data from mass-balance studies.

Summary Statistic	Amherst, MA (n = 64) Calabrese et al., 1989; Stanek and Calabrese, 1995b					Davis et al., 1990	Anaconda, MT (n = 64) Calabrese et al., 1997a; Stanek and Calabrese, 2000; Stanek et al., 2001a			
	Median Al, Si, Ti	Median[a] Top 4	Daily[b] Mean, 1+	Latent[c] Al, Si, Y	Empirical[d] Al, Si, Y	Median Al, Si, Ti	Median[e] Top 4	Daily[b] Mean, 1+	365-day average[h]	BLUP[i]
N	128[f]	128[f]	440[g]	391[g]	391[g]	101[f]	64[f]	427[g]	427[g]	64[f]
Min	< 0	< 0	< 0	0	0	< 0	< 0	< 0	< 0	< 0
Max	11,874	11,415	7,703	470	745	905	380	219	165	137
Mean	147	132	179	20	26	69	7	31	23	na
SD	1,048	1,006	na	26	47	146	75	56	na	na
Percentile										
5th	< 0	< 0	na	2	2	< 0	< 0	< 0	< 0	< 0
10th	< 0	< 0	na	4	3	< 0	< 0	< 0	< 0	2
25th	6	9	10	7	6	15	< 0	< 0	< 0	12
50th	30	33	45	12	13	44	< 0	17	13	25
75th	72	72	88	24	28	116	27	53	40	42
90th	188	110	186	43	56	210	73	111	83	75
95th	253	154	208	60	89	246	160	141	106	91

[a] Best Tracer Method; median of best 4 of 8 tracers (i.e., 4 lowest F/S ratios) for a given subject-week (Table 6, Stanek and Calabrese, 1995a).
[b] Daily Estimate Method; mean of subject-day estimates for 1 to 8 days, where each day includes at least one (1+) trace element (Table 6, Stanek and Calabrese, 1995b; Table 2, Stanek and Calabrese, 2000).
[c] Latent distribution for tracers (Al, Si, and Y); mean (2.5) and variance (0.89) of subject-day log (soil ingestion) fit to a lognormal distribution and randomly sampled 2000 times (Stanek, 1996, p.883).
[d] Empirical distribution for tracers (Al, Si, and Y); combines between-subject variance (latent variance divided by the number of subject-day estimates for each child (Stanek, 1996). Empirical distribution estimated as the sum of 2000 random samples from the latent and respon distribution, see footnote c) and within-subject variance (response error distribution - parameters fit to lognormal PDF {mean=0, variance=0.47}). Response error variance calculated as the mean of the within-subject se error distributions.
[e] Best Tracer Method; median of best 4 of 5 tracers (i.e., lowest F/S ratios) for a given subject-week (Table 13, Calabrese et al., 1997a).
[f] Number of subject-weeks represented by summary statistics.
[g] Number of subject-days represented by summary statistics.
[h] Extrapolation to 365-day average us variance components for subjects, days, and error - represented by a "shrinkage constant", yields 25 % lower values (e.g., 95th percentile reduces from 141 mg/day to 106 mg/day) (Stanek and Calabrese, 2000; p. 632, last paragraph).
[i] Stanek et al. (2001a, Table 3) and reanalysis of Stanek and Calabrese (1999) results by T. Schulz (Table 1) based on best linear unbiased predictors (BLUP) and small sample variance for subject-days.

A lognormal distribution with an arithmetic mean of 47.5 mg/day and standard deviation of 112 mg/day was fit to the percentile data using @Risk's Best Fit software (version 3.1). A tabular and graphical summary of the distribution is presented below the "Probability Distribution" section above. The RME point estimate recommended for children (EPA, 1991) of 200 mg/day is approximately the 96th percentile of this distribution. The lognormal distribution is bounded at 0 by definition, but has an infinite right tail. Given the importance of the soil ingestion rate variable in risk assessment, it is prudent to impose a upper truncation limit so that each iteration of the Monte Carlo simulation yields plausible results. The choice of an upper truncation limit is a professional judgment that weighs the confidence in the empirical data, the skew ness of the probability distribution fit to the data, and a rule of thumb to avoid overly truncating the distribution (i.e., select values that remove less than 1% of the distribution). For this analysis, an upper truncation limit of 1000 mg/day was chosen. This value is the 99.8th percentile of the distribution, and therefore constrains only 0.2% of the values.

Uncertainties in the Probability Distribution

There are multiple sources of uncertainty associated with the PDF developed to characterize interindividual variability in childhood soil ingestion rates. A comprehensive summary of potential biasing factors is given by Stanek et al. (2001b):

- Determining trace element concentrations in non-soil sources;
- Estimating gastro-intestinal transit time from food to fecal samples;
- Implementing exclusion criteria to remove unreliable daily estimates for certain tracer elements;
- Inconsistency among tracer elements in daily estimates;
- Assuming that intra-individual variability is characterized by a lognormal distribution, and that all individuals exhibit the same intra-individual variability;
- Selecting a maximum value for truncating the PDF that characterizes inter-individual variability

Selection of a Single Data Set

Multiple studies have been conducted on different study populations, including Anaconda, Amherst, and Washington State. As discussed above, the Anaconda study is considered to be more representative of the variability in soil ingestion rates among children that may be exposed in a residential scenario at Rocky Flats. It may be tempting to combine the data sets in order to increase the sample size and capture the "heterogeneity" among subpopulations of children in different locations. Given the number of differences in study design, data analysis, and population characteristics, it is not appropriate to combine the data for purposes of characterizing variability in soil ingestion rates. The different data sets do provide a measure of uncertainty, and it might be of interest to develop separate PDFs for each data set, for example. This level of quantitative uncertainty analysis is beyond the scope of this appendix.

Uncertainty Due to Model Time Step

A model time step is essentially an averaging time - it refers to the time period represented by a random value selected from a probability distribution. For most Monte Carlo models, a single random value is selected to represent a long-term average value. For example, for a single iteration of the model (representing a hypothetical child), a random value may be selected from the EDF in order to represent the average daily ingestion rate over 7 years. This is a simplifying assumption given the lack of longitudinal data on ingestion rates among individuals. An alternative would be to represent the 7-year average value by selecting 7 random year values, essentially simulating an individual's exposures over time. In general, distributions based on estimates of short-term surveys will tend to overestimate the variability in long-term average values. Until repeat measures are used to estimate ingestion rates among a population, intraindividual variability will remain an unquantifiable source of uncertainty.

The importance of the model time step assumption can be explored. Explicit model time steps can be employed to simulate an individual's exposures over time. For example, Stanek (1996) apply an annual time step because they assume that the empirical distribution described above represents interindividual variability over a 1-year period (i.e., a single random sample from this distribution represents the average IR_{soil} for an individual for the year). According to the central limit theorem, the standard deviation of the sample distribution is inversely proportional to the square root of n. Thus, decreasing the time step from one year to one month would increase the number of random samples needed to estimate the average annual ingestion rate, and effectively reduce the standard deviation of the distribution by a factor of approximately 3.5 (Goodrum et al., 1996). The effect that changing the model time step has on the distribution of IR_{soil} is summarized in FigA-4.

Several alternative approaches to simulating intraindividual variability could be explored, but were not in this analysis. For example, the method suggested by Stanek (1996) could be used to derive the response error variance of the best subject-day estimates of IR_{soil} given by the Daily Estimate Method. The resulting empirical distribution could be considered a measure of both the latent distribution and short-term variability in IR_{soil}. The model time step could then be used to explore the effect of uncertainty in extrapolating distributions over different time intervals. Another approach would be to auto correlate random samples by constraining the sample space to a percentile range of the cumulative PDF. For example, if an individual was assumed to have a high latent exposure (e.g., > 88 mg/day, the upper quartile of the IR_{soil} PDF), each consecutive random value could be weighted to the upper quartile (i.e., >75th percentile) of the distribution. This approach would simulate both the underlying, latent distribution (i.e., relatively high IR_{soil}), as well as the stochastic, short-term variability in average ingestion rates for each consecutive time step (i.e., between 88 and 7,000 mg/day).

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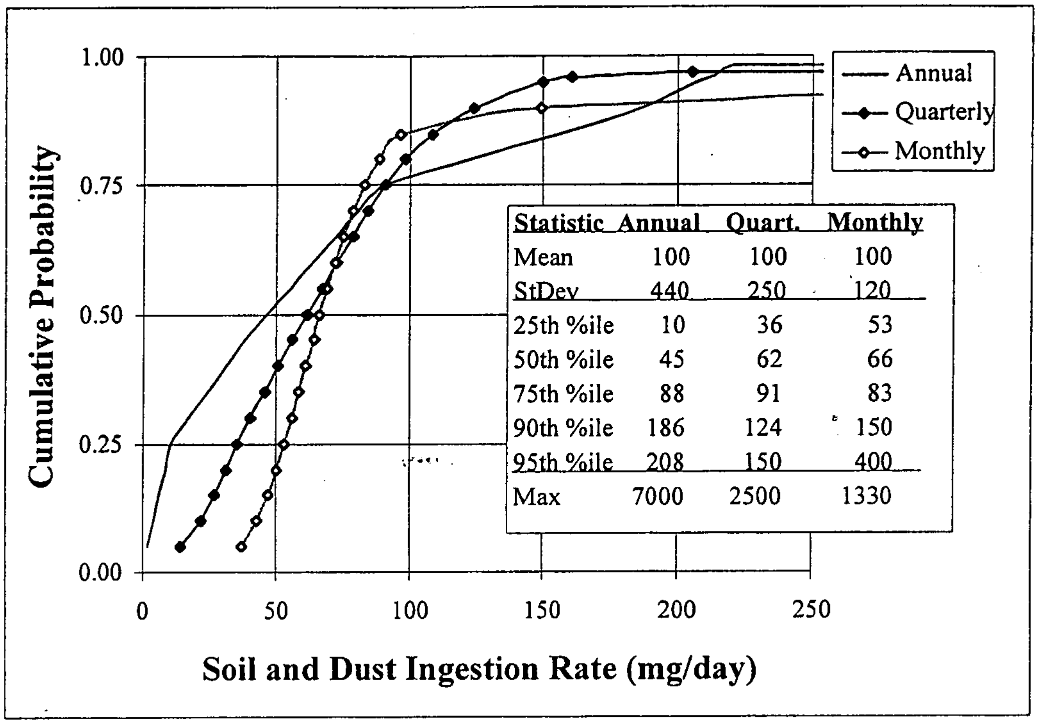

Figure A-4 Cumulative distributions of soil and dust ingestion rate based on different model time steps using Monte Carlo simulations of n = 5,000 iterations and the Amherst cohort (Calabrese, 1989).

Soil Ingestion Rate for Adults (ages 7+ years)

The soil ingestion rate variable represents the average daily mass of soil or dust that enters the human GI tract. For adults, soil ingestion is thought to reflect a combination of direct ingestion from materials placed in the mouth (e.g., hands, food, cigarettes) or indirectly via inhalation when larger particles are transferred from the upper respiratory tract to the mouth (via mucociliary transport) and swallowed.

It is generally accepted that daily activities patterns may be an important factor affecting ingestion rates. EPA Risk Assessment Guidance (U.S. EPA, 1991) differentiates between soil or dust "contact intensive" activities, in which adults are in heavy contact with soils and dusts on a regular basis (e.g., construction worker), and "non-contact intensive" activities such as the typical homeowner, office worker, or professional. However, very little data are available from which to quantify soil ingestion rates among adults for either category of activities. Therefore, the estimate for soil ingestion rate discussed below is considered to be equally applicable for each of the residential/occupational land use scenarios considered in the Rocky Flats risk assessment.

Probability Distribution

For this analysis, it was determined that insufficient data existed to develop a probability distribution for purposes of calculating risks and remediation goals. Therefore, a point estimate of 100 mg/day is used in the analysis, based on the value recommend by EPA (1991) for adult populations in residential and agricultural scenarios.

For purposes of sensitivity analysis, it may still be useful to develop a probability distribution in order to evaluate the influence of this variable on the risk distribution. If a Monte Carlo sensitivity analysis is run, the following probability distribution is recommended for use in risk equations that are based on U.S. EPA Risk Assessment Guidance in order to characterize *interindividual* variability in adult soil ingestion rate:

IRs_adult ~ Uniform (30, 100) mg/day

The uniform distribution is defined by two parameters:

- minimum 30 mg/day
- maximum 100 mg/day

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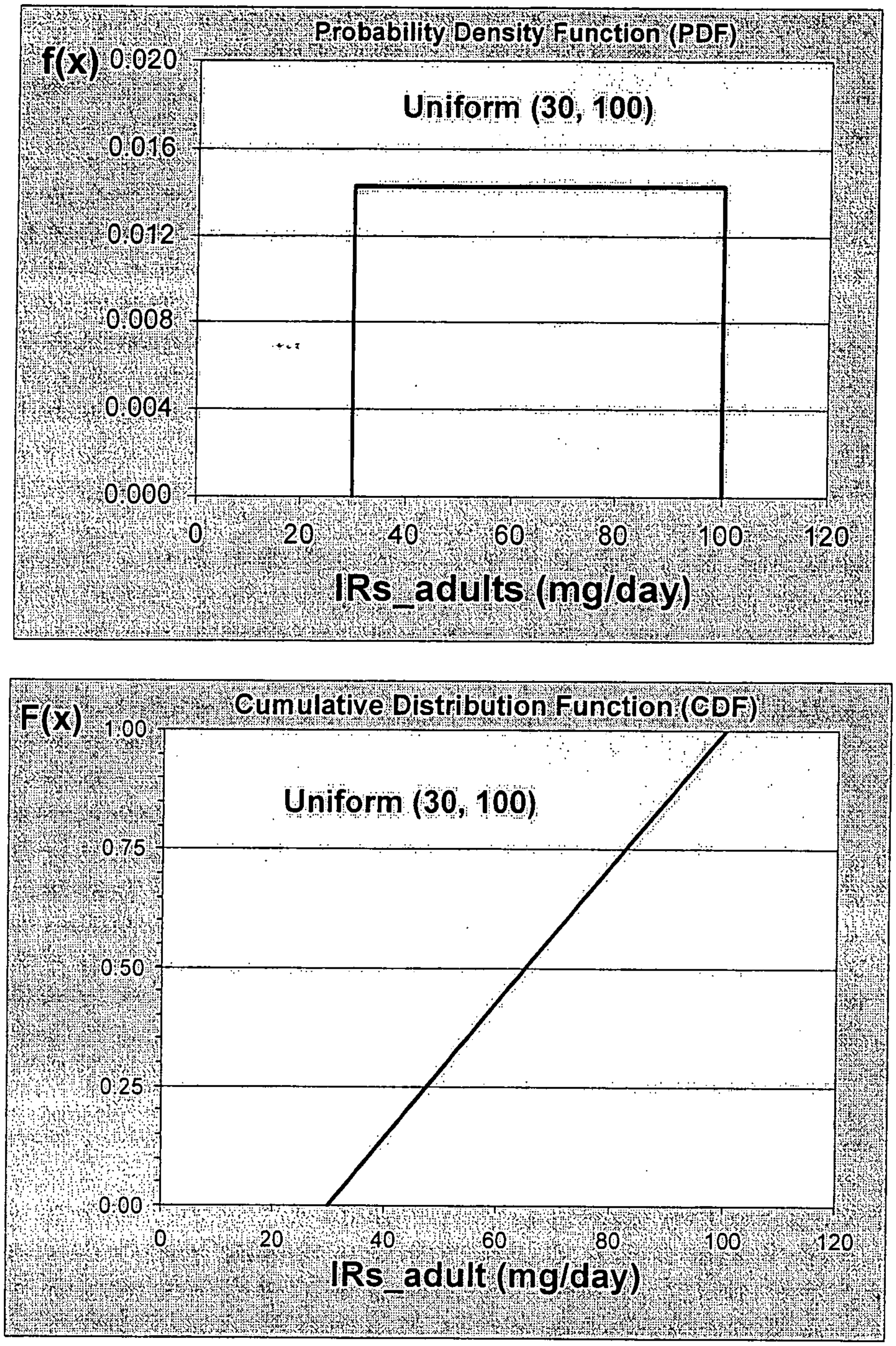

Figure A-5 Probability density function (PDF) and cumulative distribution function (CDF) views of the uniform distribution for adult soil ingestion rate (mg/day).

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For the RESRAD model, the same point estimate can be used by converting the units from (mg/day) to (g/year):

- point estimate   100 mg/day x 0.001 g/mg x 350 day/yr = **35 g/year**

Similarly, a probability distribution used in a sensitivity analysis would have the following parameters:

- minimum   30 mg/day x 0.001 g/mg x 350 day/yr = 10.5 g/year
- maximum   100 mg/day x 0.001 g/mg x 350 day/yr = 35 g/year

Therefore, the equivalent distribution for use in RESRAD is:

IRs_adult ~ Uniform (10.5, 35) g/year

Uncertainties in the Probability Distribution

The limited data available on soil ingestion rates in adults poses a challenge when attempting to develop a probability distribution that characterizes interindividual variability. The following discussion provides highlights of the available empirical data, and an overview of the reasoning used in developing the recommended distribution.

Calabrese et al., 1990 Study for Adult Soil Ingestion Rate

Empirical data on adult soil ingestion rates are available from a single study (Calabrese et al., 1990), conducted concurrently with a study of childhood soil ingestion rates in Amherst, MA. The purpose of the adult study was to verify the tracer mass balance methodology used in the child study, rather than to investigate the amount of soil normally ingested by adults. Nevertheless, as indicated by the authors, it does offer an estimate of the amount of soil ingested by the six adult subjects in the study over a period of three consecutive days for each of three weeks.

A more detailed summary of the best tracer methodology used to estimate soil ingestion rates is given in the discussion on the probability distribution developed to characterize soil ingestion rates in children in this Appendix A. Stanek and Calabrese (1995) recommend estimating a distribution of soil ingestion rates from this type of study based on the median of the four best tracers for each subject week. On the basis of percent recoveries, the four best tracers for this study were determined to be Al, Si, Y, and Zr. Results of the study reported by week and tracer are given in Table A-5.

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Table A-5. Calabrese et al. 1990 (Table 7, p. 93) study results by week and tracer element based on median Amherst soil concentrations [mean / median for n = 6 subjects].

Study Week	Soil Ingestion (mg/day) by Tracer [mean / median]			
	Al	Si	Y	Zr
1	110 / 60	30 / 31	63 / 44	134 / 124
2	98 / 85	14 / 15	21 / 35	58 / 65
3	28 / 66	-23 / -27	67 / 60	-74 / -144

The data may also be grouped by individual and tracer element, and averaged across all three weeks, as shown in Table A2. Corresponding estimates for each of the 6 individuals are given in Figure A2.

For the three weeks of data (Table A-5), the minimum, non-negative average soil ingestion rate (i.e., averaged across all six subjects) is given by Si (14 mg/day), while the maximum is by given Zr (134 mg/day).

For the six subjects (Table A-6), the minimum, non-negative average soil ingestion rate (i.e., averaged across all three weeks) is given by Al (1 mg/day), while the maximum is given by Zr (216 mg/day). If the estimates are further averaged across individuals, the mean soil ingestion rate ranges from 5 to 33 mg/day, while the median ranges from -4 to 65 mg/day.

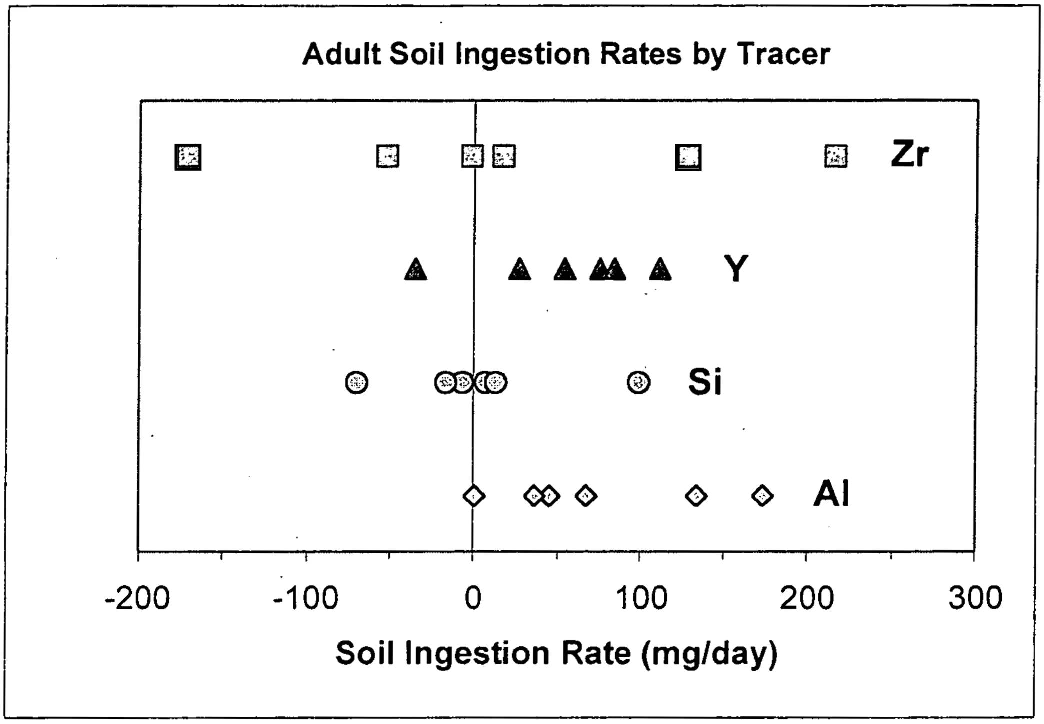
Negative ingestion rates occur due to complexities in the tracer mass balance methodology, such as the assumed transit time in the GI tract and the non-soil sources of tracer elements. The tracer element with the most variable results (given by the reported standard deviation in Table A2) is Zr (SD = 141 mg/day), while the least variable is Si (SD = 55 mg/day). The distribution of ingestion rates by individual is more clearly shown in Figure A2.

Table A-6. Calabrese et al. 1990 (Table 8, p. 94) study results by individual and tracer element based on median Amherst soil concentrations [for n = 3 weeks]. Also see Figure A2.

Subject Statistics	Soil Ingestion (mg/day) by Tracer			
	Al	Si*	Y*	Zr*
minimum	1	7	27	17
maximum	173	99	111	216
mean	77	5	53	33
median	57	1	65	-4
standard dev.	65	55	51	141

*Statistics include negative estimates; 3/6 estimates were negative for Si and Zr while 1/6 was negative for Y, as shown in Figure A-6.

Figure A-6 Calabrese et al. (1990) results for 4 best tracers showing 3-week average estimates for each of n=6 individuals. Summary statistics across individuals are given in Table A2.

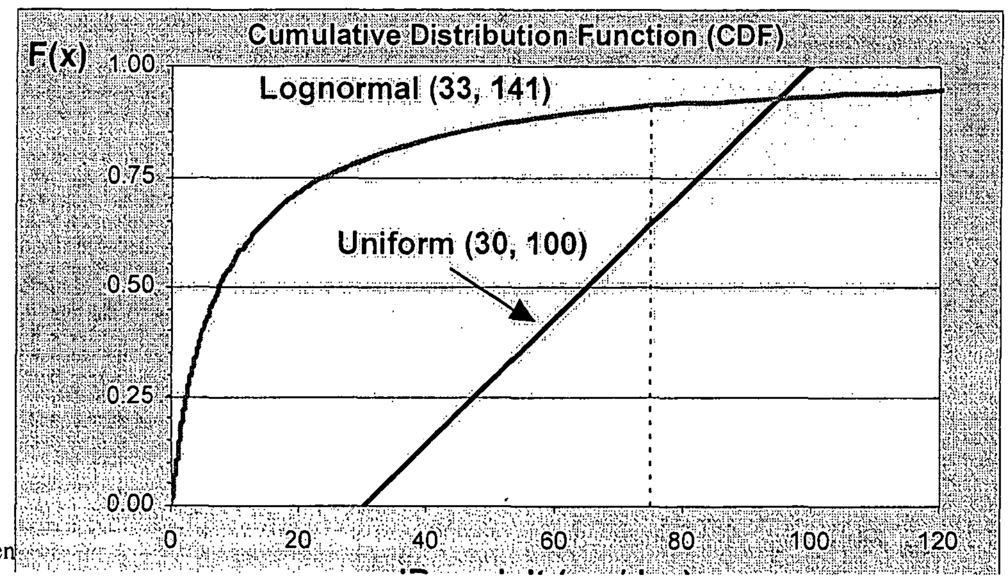


Basis for Uniform (30, 100)

Based on the limited empirical data (1 study with n=6), no attempt was made to evaluate different plausible distributions. The available information does support a plausible minimum and maximum value. For example, a minimum of as low as 1 mg/day and maximum as high as 216 mg/day are plausible. Both estimates are based on an average of daily values for 3 separate weeks; since the short term data are intended to represent a long term average, a reasonable assumption is that these estimates are more extreme than may be necessary. This is because most individuals will tend to experience a range of conditions over a long time period (e.g., years), and very high (or low) estimates measured during 1 week are likely to be offset by different exposures the next. This process is sometimes referred to as "averaging towards the mean", and presents a major challenge in applying short term survey data to risk assessments. A range of 30 to 100 mg/day was selected based on professional judgment. The minimum of 30 mg/day is greater than 45% of the subject/tracer measurements, and the maximum of 100 mg/day is less than 20% of the measurements; therefore, while it constrains the short term data to allow for applications to long-term exposures, this range is weighted toward higher ingestion rates.

Given a plausible range, but no further information regarding the shape or spread of the distribution (e.g., mean, standard deviation), a uniform distribution was selected for use in a sensitivity analysis. A uniform distribution gives equal probabilities to any value within the range. This can be contrasted with a normal or lognormal distribution, for which values at the tails of the distribution are much less likely than those nearer to the mean or median. For example, if a lognormal distribution was selected with a mean of 33 mg/day and standard deviation of 141 mg/day (loosely based on the tracer element Zr), an ingestion rate of 75 mg/day would be the 91st percentile of the distribution (i.e., less than 10 % of values are expected to be greater than 75), whereas with the uniform distribution, nearly one third of the values are expected to be greater than 75 mg/day. Figure A-7 clearly illustrates this concept. Given the available information, the use of the uniform distribution is considered to be a more protective choice than other distributions because more weight (probability) is given to higher ingestion rates. For example, 75 mg/day is approximately the 65th percentile of the uniform, but the 91st percentile of the lognormal. The uniform is truncated at the maximum value of 100 mg/day, whereas the lognormal is untruncated at the high end and will yield ingestion rates greater than 100 mg/day (7% of the time). However, estimates at these high ends are uncertain given the small number of study subjects and variability among different tracer elements.

Figure A-7. Comparison of the Uniform (30, 100) distribution and the Lognormal (33, 141) distribution showing how higher ingestion rates are more likely with the use of the uniform.



The use of a probability distribution for purposes of risk estimation and calculations of PRSALs is not recommended due to the limited data available. However, for purposes of a sensitivity analysis, to explore the influence of this variable on the risk estimate, a uniform distribution may be used. Use of the uniform PDF is a judgment call that requires consideration of two key factors: 1) the objectives of the Monte Carlo modeling approach, and 2) the representativeness, quantity, and quality of the available data. For this analysis, the ultimate goal is to use quantitative information on variability and uncertainty in exposure to help inform the risk management decisions at Rocky Flats.

An important component of a Monte Carlo simulation is the sensitivity analysis, which can help to focus the interpretation of the risk distributions on the key variables. Variables that are represented by point estimates are essentially excluded from the sensitivity analysis because they do not contribute to variability in the risk estimates. Secondly, while the empirical data are sparse, it is reasonable to assume that the study was appropriately conducted and that the subjects are representative surrogates for a larger population of adults. In other words, the main deficiency is that there are too few measurements to evaluate additional distributions with any confidence. The selection of a uniform distribution reflects a balance between the available data, and the information that can be provided for the risk management decision by allowing adult soil ingestion rate to contribute to the overall sensitivity analysis. In addition, the parameters selected for the uniform distribution (min, max), while largely based on judgment, were informed by the available data and do reflect an effort to yield higher soil ingestion rates in the risk model than would otherwise have been obtained with selections of other probability distributions.

Rural Resident: Vegetable, Fruit, and Grain Ingestion Rate

For the rural resident land use scenario, one potential exposure pathway is the consumption of plants grown in a family garden. Home-grown commodities considered in this analysis include vegetables, fruit, and grain. The total amount of these foods ingested on an average day may be thought of as the sum of the home-grown foods plus the foods purchased from the market. The ideal data set for estimating *interindividual* variability (between individuals) in average daily ingestion rates among children and adults would include information on factors described below (see Table A-7). These factors may provide a benchmark for determining the representativeness of ingestion rate data for purposes of a risk assessment for the rural resident exposure scenario.

The USDA Nationwide Food Consumption Survey (NFCS) is the largest publicly available source of information on food consumption habits in the United States. Data from the most recent survey conducted in 1987-1988, which included approximately 4,300 households and 10,000 individuals, have been summarized in the U.S. EPA Exposure Factors Handbook (EFH) (EPA, 1997). Respondents estimated intakes over a 1-week period. These data summaries were used to develop probability distributions to characterize variability in average daily ingestion rates of vegetable and fruit, as described in detail below.

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Table A-7 Examples of information on vegetable, fruit, and grain ingestion rates that would provide high confidence in the risk estimates for the residential scenario.

Item	Information	Importance for Risk Assessment
1	Fraction homegrown	Risk assessments generally focus on exposures resulting from on-site contamination. Foods grown on site are more relevant than foods purchased from the market. If fraction homegrown is not considered, risks will generally be overestimated for most populations.
2	Consumers only	The target population for the risk assessment is individuals who consume vegetables, fruit, and/or grain. Individuals that do not consume these commodities in general (or during the short study period of the survey) would be included in "per capita" estimates, which would be lower than "consumer only" estimates. Estimates for consumers only would be more representative.
3	Season-specific estimates	Dietary patterns may shift seasonally depending on the availability of certain commodities, especially when the risk assessment focuses on home-grown (rather than store-bought) items. Long-term estimates of average daily ingestion rates would be biased if they did not account for seasonal variability. Seasonal ingestion rates are likely to vary by region (see Item 5), depending on the climate, length of the growing season, and availability of alternative foods from the same category (e.g., fruit and vegetables).
4	Short-term and long-term average daily rates	National survey data typically reflect dietary patterns over a short period of time (e.g., 1 week), whereas a risk assessment generally focuses on long-term exposures, especially for chronic health endpoints like cancer. In the absence of data providing estimates from a subpopulation over multiple time intervals, reasonable assumptions are needed to extrapolate to longer time periods.
5	Region-specific estimates	Estimates based on a subset of the data representative of a region or county can indirectly account for both environmental factors (e.g., climate and soil type) and demographic factors (e.g., race, ethnicity, economic status, and degree of urbanization). Data grouped into the West are most relevant to sites in Colorado.
6	Age-specific estimates	For the Rocky Flats assessment, residents are assumed to begin exposures during childhood (≤7 years) and continue through adulthood (> 7 years).
7	Relevant Subgroups of Commodities	Some plants, such as leafy vegetables, may be a source of exposure either due to uptake of radionuclides from soil or deposition of contaminated dusts on the leafy surfaces. By contrast, foliar deposition is not expected to contribute to exposures for non-leafy vegetables (e.g., carrots). Ingestion rates that distinguish leafy from non-leafy vegetable consumption are preferred in the risk assessment.

The USDA Continuing Survey of Food Intakes by Individuals (CSFII), together with NFCS, is the primary source of information on ingestion rates of grain products in the United States. Data from the 1989-1991 CSFII survey, which is considered to the key study for intake rates of grain products (EPA, 1997), was used to develop probability distributions to characterize variability in average daily ingestion rates of total grain, as described below. Respondents estimated intakes over a 3-day period.

Table A-8 summarizes the characteristics of the available data on average daily ingestion rates of vegetables, fruit, and grain based on the factors listed in Table A-7. The summary data on vegetables and fruit contain many of the characteristics relevant for application to risk assessment, with the exception of a distinction between leafy and non-leafy vegetables (Item 7). Data on grain ingestion rates are also very comprehensive, but do not provide any information regarding the home-grown fraction (Item 1)[1]. In addition, a general observation for all of the survey data is that there is uncertainty in applying information based on short term dietary patterns (i.e., days or weeks) to estimate long-term ingestion rates (e.g., years) among the U.S. population.

Table A-8. Information on vegetable, fruit, and grain ingestion rates from Table A-7 that are reported by U.S. EPA Exposure Factors Handbook (EFH) (EPA, 1997).

Item	Information	Vegetable	Fruit	Grain
1	Fraction homegrown	X	X	
2	Consumers only	X	X	X
3	Season-specific estimates	X	X	
4	Short-term and long-term average daily rates			
5	Region-specific estimates	X	X	X
6	Age-specific estimates	X	X	X
7	Portions of plant expected to have different concentrations[1]			

[1] Concentrations of elements in plants may vary depending on whether they grow above or below ground. For example, vegetables may be divided into leafy and non-leafy (i.e., root) categories.

[1] Two basic approaches can be used to quantify exposures from homegrown commodities: 1) Estimate the total consumption rates of each food category and multiply this value by the estimated home-grown fractions of each category; or 2) Use summary statistics for home-grown commodities. The first approach was used for grain, in the absence of summary data on home-grown grain ingestion. The second approach was used to develop probability distributions for vegetables and fruit.

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Probability Distribution

For this analysis, probability distributions were generated from the empirical distribution functions reported in the U.S. EPA Exposure Factors Handbook (EFH) (EPA, 1997). For each data set, 9 percentile values were reported (ranging from 1st to 99th) as well a the mean and maximum. In addition, the intake rates were normalized to body weight and expressed in units of grams of food per kilogram body weight per day (g/kg-day). Despite the large sample sizes of the national surveys, the maximum ingestion rate reported from the survey may not represent a plausible maximum ingestion rate for the population. Table A-9 presents the data used in this analysis, both on a g/kg-day basis and converted to g/day assuming 15 kg body weight for children and 70 kg body weight for adults.

Table A-9. Empirical distributions of intake rates for vegetables, fruit, and grain as reported by the U.S. EPA Exposure Factors Handbook (EPA, 1997) in g/kg-day, and converted to kg/yr

Percentile of ECDF	Vegetables			Fruit			Grain		
	Table* 13-33	kg/yr child	kg/yr adult	Table* 13-33	kg/yr child	kg/yr adult	Table* 12-1	kg/yr child	kg/yr adult
0.01	1.80E-03	0.01	0.04	5.50E-04	0.00	0.01	0	0.0	0.0
0.05	1.91E-02	0.10	0.47	5.66E-02	0.30	1.39	0.69	3.6	16.9
0.10	3.83E-02	0.20	0.94	8.82E-02	0.46	2.16	1.13	5.9	27.7
0.25	1.14E-01	0.60	2.79	2.87E-01	1.51	7.03	1.92	10.1	47.0
0.50	4.92E-01	2.58	12.05	6.88E-01	3.61	16.86	3.13	16.4	76.7
0.75	1.46E+00	7.67	35.77	1.81E+00	9.50	44.35	5.03	26.4	123.2
0.90	2.99E+00	15.70	73.26	4.75E+00	24.94	116.38	7.98	41.9	195.5
0.95	5.04E+00	26.46	123.48	8.54E+00	44.84	209.23	10.90	57.2	267.1
0.99	8.91E+00	46.78	218.30	1.45E+01	76.13	355.25	19.50	102.4	477.8
1.00	1.12E+01	58.80	274.40	1.84E+01	96.60	450.80	25.89	135.9	634.3

Unit conversion: kg/yr = g/kg-day x average body weight x 0.001 kg/g x 350 d/year; body weights for children and adults were assumed to be 15 kg and 70 kg, respectively.

* Exposure Factors Handbook (1997)

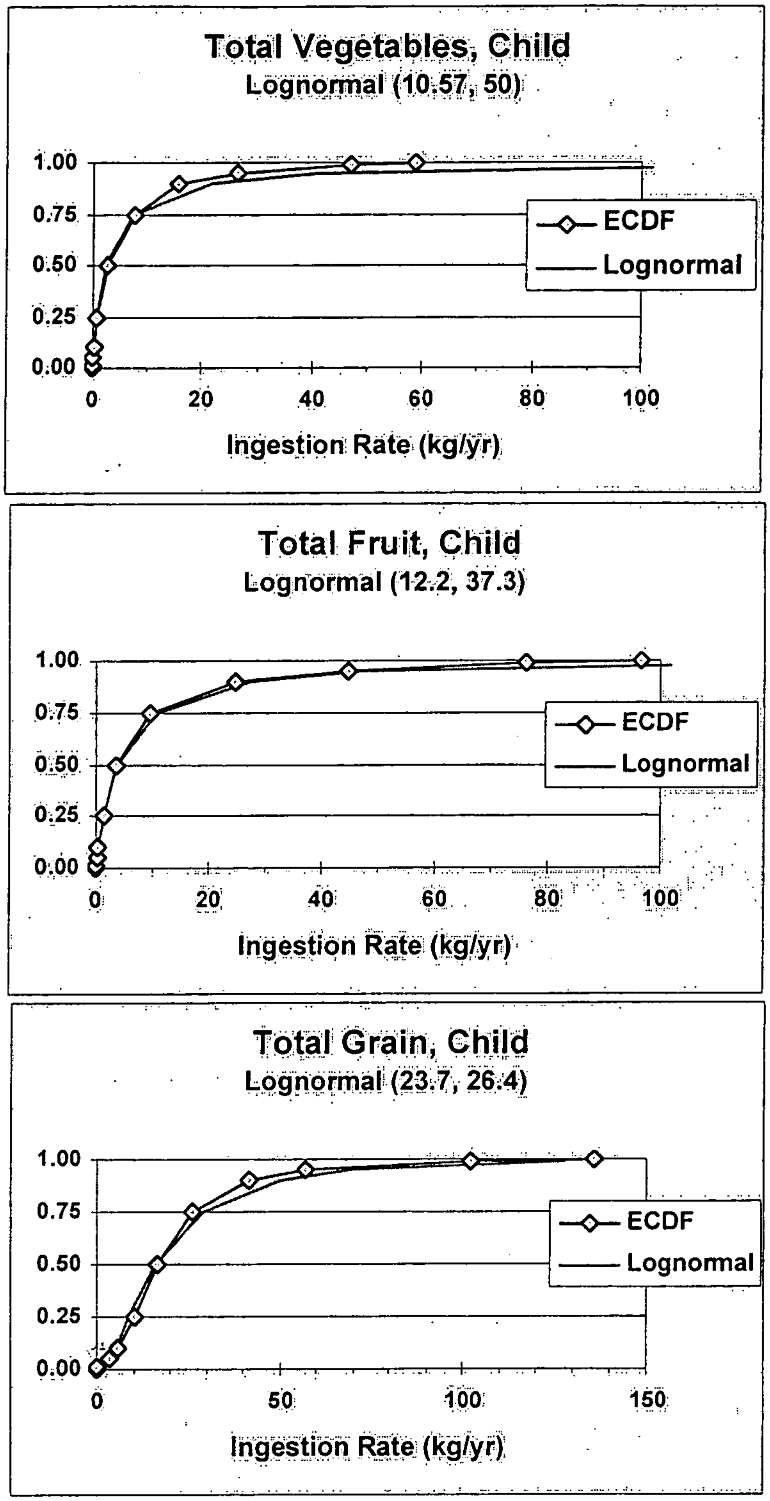

Figure A-8 Comparison of empirical and lognormal cumulative distribution functions (CDFs) for ingestion rates of vegetable, fruit and grain by children.

IR_food ~ Lognormal (mean, SD) kg/year

The lognormal distribution is defined by two parameters. Values for childhood ingestion rate of total vegetables are given below as an example:

- arithmetic mean 10.57 kg/yr
- standard deviation 50.00 kg/yr

For this analysis, truncation limits were not applied. By definition, the lognormal distribution is truncated at the low end at 0 (i.e., non-negative values), which is a reasonable lower limit for this variable. The upper truncation limits could be specified

Empirical data can be used directly in a probabilistic risk assessment by specifying an empirical cumulative distribution function (ECDF). Alternatively, the percentile values can be fit to a probability distribution. Several continuous distributions were evaluated for this analysis based on visual inspection and goodness-of-fit (GoF) statistics using @Risk (Palisades Corp.). Although @Risk does provide GoF statistics, these should be interpreted with caution given that GoF techniques are typically applied to raw data values rather than percentile data. Nevertheless, the Chi-Square and Kolomogorov-Smirnoff test statistics provide an additional metric for evaluating the relative fits of the observed percentile data to F(x), the percentiles of the hypothesized distribution. Lognormal distributions provided an adequate fit for most of the summary data. Results of graphical analysis and maximum likelihood parameter estimates are given below. Table A-10 summarizes the distributions and parameter estimates used in the risk assessment.

Table A-10. Summary of parameter values for lognormal distributions used to characterize variability in vegetable, fruit, and grain ingestion rates

Average Daily Ingestion Rates by Plant and Age Group			
Plant	**Child (< 7 yrs)**	**Adult (7+ yrs)**	**Age-Adjusted[1]**
Vegetable, total	[10.57, 50]	[50, 240]	
Vegetable, leafy	[1.57, 7.45]	[7.45, 35.76]	[6.3, 28.6]
Vegetable, non-leafy	[9.00, 42.55]	[42.55, 204.24]	[35.8, 163.6]
Fruit, total	[12.2, 37.3]	[57, 174]	
Grain, total	[23.65, 26.4]	[110, 123]	
Non-leafy Vegetable + Fruit + Grain	[21.4, 56.6]	[100.7, 268.3]	[84.8, 214.9]

[1] Age-adjusted = (6/30) x IR for child + (24/30) x IR for adult

Uncertainties in the Probability Distribution

The summary tables given in EFH reflect a number of simplifying assumptions and statistical methods that may be important to understand in order to characterize the uncertainties associated with this exposure pathway. These are briefly described below.

Per capita vs. consumers only. Consumers are defined as members of a household who reported consumption of the food item/group of interest during a the survey period. *Per capita* estimates reflect the combination of respondents who reported intakes during the study period (i.e., consumers) and individuals who may consume a commodity in the future.

Age-specific estimates based on body weight. Data are reported on a body weight-normalized basis (grams of food per kg body weight per day). To convert to an intake rate (g/day) for the risk assessment, it is necessary to multiply values by body weight (kg). For the Rocky Flats risk assessment, the target population is divided into two age groups - children and adults. As summarized in EFH, the average body weight for children ages 6 months to years is approximately 15 kg (EPA, 1997, Table 7-3) and adults ages 18 to 75 years is approximately 70 kg (EPA, 1997, Table 7-2). These weights were applied to the data to generate age-specific distributions. According to EFH (EPA, 1997, pages 13-7 to 13-9), the average body weight of respondents (children and adults combined) was approximately 60 kg. If an exposure duration of 30 years is used in a risk assessment, with 6 years representative of children and 24 years representative of adults, the mean body weights used in this analysis match this result very closely, as shown below:

Extrapolation to long-term estimates. The percentiles of the average daily intake were converted from the short time interval of 3-7 days to a long-term average by averaging the corresponding percentiles of each of four seasonal distributions for the same region (EPA, 1997, p. 13-3). This approach reflects an assumption that each individual consumes at the same regional percentile levels for each week of a season, and each season of the year. For example, an individual whose combined ingestion rate of vegetable, fruit, and grain is the 90th percentile for one week in the summer in the West, would be assumed to also consume at the 90th percentile for each week and season.

Summation of ingestion rates by individual Several methods may be used to estimate the average daily ingestion rates for multiple commodities (vegetable + fruit + grain). The preferred method would account for potential correlations for a given individual in their dietary preferences and choices of types of foods grown at home. This correlation would be maintained if the summation were estimated at the level of the individual records from the survey data, rather than pooling data from the entire sample for each commodity, and summing at the population level. In short, the average of the total ingestion rates reported by individual is more representative than the sum of the average ingestion rates reported for each commodity. Since such data were not available from EFH, the total ingestion rate was calculated by summing the distributions for each commodity.

Subpopulations for Vegetable and Fruit ingestion rate. Table 13-33 in EFH (EPA, 1997) was used to derive probability distributions for average daily ingestion rates of total vegetables and

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fruit (i.e., seasonally adjusted, consumer only, home-grown, West region, total vegetables, total fruit).

Subpopulations for Grain ingestion rate. Table 12-1 in EFH (EPA, 1997) was used to derive a probability distribution for average daily grain ingestion rate (per capita, West region, total grains including mixtures). Data could be selected by age group, *or* by region for all ages combined, but there are no regional age-specific data. For this analysis, distributions are based on data by region (i.e., West) and average body weights for children and adults are used to derive age-specific distributions. It is unclear how variability in ingestion rates among children compare with variability for adults.

Homegrown fraction for Grain. There are no data available on home-grown fraction of total grain ingestion rate. The home-grown fraction would represent the family that harvests the grain at home in order to prepare grain products such as flour for breads. This fraction is expected to be relatively low, as compared with home-grown fractions for vegetables (17% for gardeners, 31% for farmers) and fruit (10% for gardeners, 16% for farmers) (EPA, 1997). It was assumed that only 1 percent of the population grows and prepares grain products at home.

Seasonal variability for grains. Seasonal patterns are thought to be minor source of variability in grain consumption (EPA, 1997, p. 12-1) because grains may be eaten on a daily basis throughout the year. Therefore, the distribution based on short-term data is considered a reasonable approximation of the long-term distribution, although it will display somewhat increased variability (EPA, 1997).

Inhalation Rate (IR_air) Rural Resident

Inhalation rate refer to the volume of air that is inhaled over a period of time. Studies of human inhalation rates have demonstrated variability associated with age, gender, weight, health status, and activity patterns (i.e., resting, walking, jogging, etc.). Although an individual's inhalation rate will vary day-to-day and week-to-week, inhalation rates used in risk assessment general describe an average daily rate (m^3/day) over a long period of time (i.e., the exposure duration). If acute exposures associated with moderate to heavy activities may be of concern, estimates of average hourly inhalation (m^3/hour) would generally be preferred over of daily averages. Average daily or hourly inhalation rates will vary between people, and it is this interindividual variability that is characterized by a probability distribution for this analysis. Short-term measurements, referred to as "minute volumes" (L/min), form the basis for long-term average ingestion rates. The literature on inhalation rates is fairly robust, and can be loosely grouped into two categories based on study methodology: 1) direct measurements using a spirometer, or 2) indirect measurements based on correlations with heart rate, energy requirements, and/or other physiological factors. Data from U.S. EPA Exposure Factors Handbook (EPA, 1997), and a subsequent publication by Allan and Richardson (1998) on 24-hour inhalation rates formed the basis for the estimates described below.

<u>Probability Distribution</u>

The following probability distribution was developed for use in probabilistic risk and RSAL calculations for the rural resident land use scenario:

- **IR_air_child ~ Lognormal (9.3, 2.9) m^3/day**
- **IR_air_adult ~ Lognormal (16.2, 3.9) m^3/day**

The lognormal distributions are defined by two parameters:

- arithmetic mean 9.3 and 16. 2 m^3/day
- standard deviation 2.9 and 3.9 m^3/day

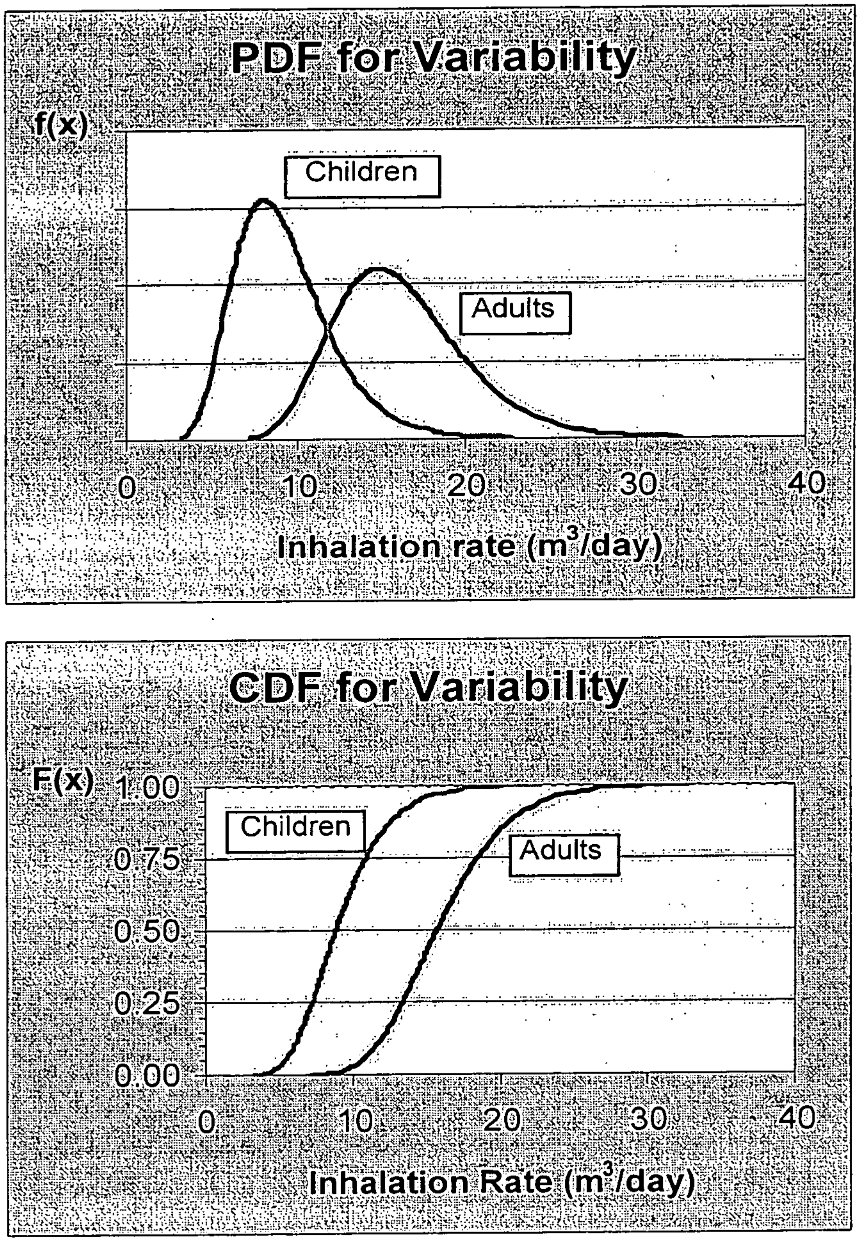

Figure A-9 Probability density function (PDF) and cumulative distribution function (CDF) views of the probability distribution for child and adult inhalation rate (m^3/day).

Uncertainties in the Probability Distribution

The U.S. EPA Exposure Factors Handbook (EFH) (U.S. EPA, 1997) provides a comprehensive summary of the available data on inhalation rates. In addition, U.S. EPA ORD recently presented recommendations for probability distributions for inhalation rates (U.S. EPA 2000).

Table A-12 summarizes some of data available from some of the key studies on inhalation rates. Variability in inhalation rates at most activity levels are generally positively skewed, with more minute volumes nearer the lower end of the reported ranges (Allan and Richardson, 1998). Since inhalation is a non-negative quantity, the literature tends to report lognormal distributions fit to the available data. Allan and Richardson provide graphical summaries of the fits, but no description of goodness-of-fit test statistics. Adult males tend to exhibit the highest inhalation rates, with an average of approximately 17.5 m^3/day. More importantly, there is remarkable consistency in estimates for both children and adults:

- estimates of average inhalation rates among toddlers and young children exhibit a range of approximately 1 m^3/day (a minimum of approximately 8.7 m^3/day to a maximum of 9.7 m^3/day.
- estimates of average inhalation rates among adults exhibit a range of approximately 6 m^3/day (11.3 - 17.5 m^3/day).
- within study groups, the interindividual variability is very low, as shown by coefficients of variation (ratio of standard deviation to the mean) of approximately 0.25.

For children (males/females combined, ages 7 months to 4 years) the available data fit a lognormal distribution with parameters (arithmetic mean, standard deviation) of [9.25, 2.57] m^3/day. For adults (males/females combined) the available data also fit a lognormal distribution [16.2, 3.86] m^3/day. These results are within the range of all reported values, as well as the values recommended by U.S. EPA for risk assessment (EPA, 1997):

Table A-11. Summary of recommended values for inhalation rates (U.S. EPA, 1997, Table 5–23).

Age Group	Inhalation Rate (m^3/day)	
	Long-term Exposure	Short-term Exposure
Child, 1-2 years	6.8	rest - 0.3 sedentary - 0.4 light activity - 1.0 moderate activity - 1.6 heavy activity - 1.9
Child, 3-5 years	8.3	
Child, 6-8 years	10.0	
Adult, 19+ years	11.3 - 15.2	rest - 0.4 sedentary - 0.5 light activity - 1.0 moderate activity - 1.6 heavy activity - 1.9
Adult Worker	hourly average - 1.3 m^3/hr hourly average, high end - 3.3 m^3/hr slow activities - 1.1 m^3/hr moderate activities - 1.5 m^3/hr heavy activities - 2.5 m^3/hr	

Table A-12 Summary of point estimates and probability distribution parameters for inhalation rates.

Point Estimate				
Population	***CTE, RME***	***Units***	***Source***	***Comments***
Child (? 6 yrs), M/F	8.7, --	m3/day	U.S. EPA, 1996	Long-term exposures for children 1-12 years
Adult (> 6 yrs), male	15.2, --	m3/day	U.S. EPA, 1996	Long-term exposures for adult males
Adult (> 6 yrs), female	11.3, --	m3/day	U.S. EPA, 1996	Long-term exposures for adult females
Outdoor worker	1.3, 3.5	m3/hr	U.S. EPA, 1996	Short-term exposures for outdoor workers, hourly average
Probability Distribution - U.S. EPA EFH				
Population	***Type***	***Parameters (m3/day)***	***Source***	***Comments***
Child (? 6 yrs), male	Lognormal[1]	9.30, 2.85	U.S. EPA, 2000 (Moya)	Based on Layton (1993) study in which inhalation rates were based on BMR and energy expenditures; children aged 3-10 years
Child (? 6 yrs), female	Lognormal[1]	8.65, 2.65	U.S. EPA, 2000	Children aged 3-10 years
Adult (> 6 yrs), male	Lognormal[1]	16.75, 5.32	U.S. EPA, 2000	Adults aged 18-30 years
Adult (> 6 yrs), female	Lognormal[1]	11.14, 5.37	U.S. EPA, 2000	Adults aged 18-30 years
Probability Distribution - Other Sources				
Population	***Type***	***Parameters (m3/day)***	***Source***	***Comments***
Child (? 6 yrs), male	Lognormal[1]	9.67, 2.67	Allan and Richardson, 1998	Study of Canadian subjects using time-activity patterns and minute volumes from USA studies; values represent 24-hr inhalation rates; male children 7 months to 4 years of age
Child (? 6 yrs), female	Lognormal[1]	8.81, 2.37	Allan and Richardson, 1998	Female children 7 months to 4 years of age
Child (? 6 yrs), M/F	Lognormal[1]	9.25, 2.57	Allan and Richardson, 1998	M/F children 7 months to 4 years of age
Adult (> 6 yrs), male	Lognormal[1]	17.54, 4.06	Allan and Richardson, 1998	Male adults 20 to 59 years of age
Adult (> 6 yrs), female	Lognormal[1]	14.89, 3.13	Allan and Richardson, 1998	Female adults 20 to 59 years of age
Adult (> 6 years), M/F	Lognormal[1]	16.2, 3.86	Allan and Richardson, 1998	M/F adults 20 to 59 years of age

[1]Lognormal distribution parameters are the arithmetic mean and standard deviation. Primary Reference: Allan, M. and Richardson, G. 1998. Probability density functions describing 24-hour inhalation rates for use in human health risk assessments. *Hum. Ecol. Risk Assess.* 4(2): 379-408.

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Inhalation Rate (IR_air) Wildlife Refuge Worker

Inhalation rates for workers will vary greatly, depending on the time spent at different levels of activity. While inhalation may be expressed on as an average daily rate (by averaging over an 8-hour workday), the basic unit of interest is the short-term average rate (e.g., minutes or hours). The Rocky Mountain Arsenal (RMA) risk assessment (reference) provides estimates of inhalation for biological workers based on a calculation of the time-weighted average breathing rates (see Section B.3.4.1.4 of RMA). These estimates formed the basis for the probability distributions used in this analysis.

Probability Distribution

The following probability distribution was developed for use in probabilistic risk and RSAL calculations for the rural resident land use scenario:

$$\mathbf{IR\_air\_wildlife \sim min + (max - min) \times Beta\ (a, b)\ \ m^3/hr}$$

The beta distributions are defined by four parameters:

- shape parameter a 1.79 m^3/hr
- shape parameter b 3.06 m^3/hr
- minimum 1.1 m^3/hr
- maximum 2.0 m^3/hr

Information on the beta distribution is provided at the end of this Section.

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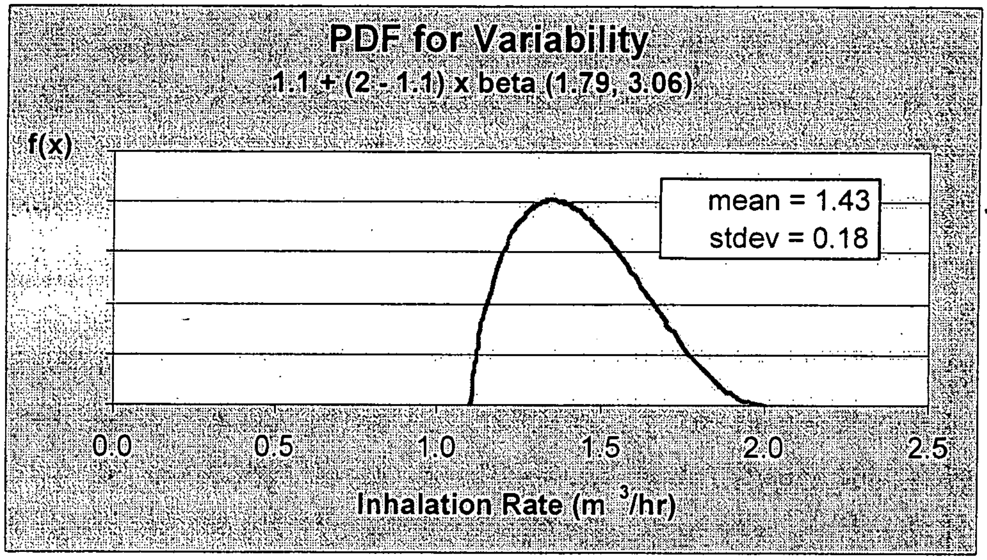
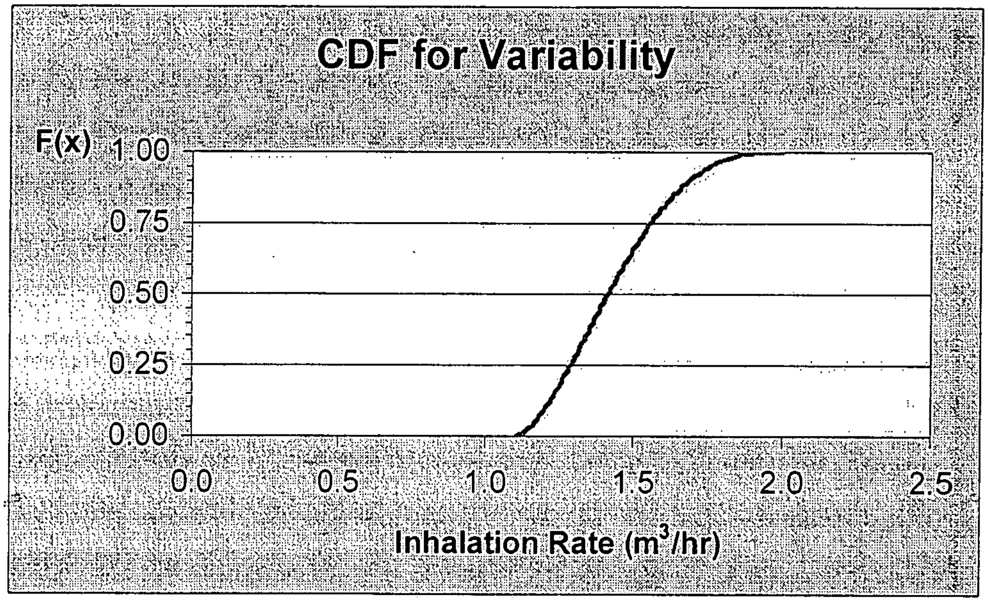
Figure A-10 Probability density function (PDF) and cumulative distribution function (CDF) views of the probability distribution for wildlife refuge worker inhalation rate (m^3/hr).





Uncertainties in the Probability Distribution

The RMA report describes the methodology use to generate the estimates of the time-weighted average breathing rates among biological workers. A brief description is given here. Activity patterns were divided into three categories based on the extent of contact with site soils:

P1 (indoor), P2 (middle), and P3 (higher)

Survey data on activity patterns among biological workers were used to develop a discrete probability distribution for the amount of time engaged in each category. In addition, three categories of breathing rates were specified:

BR (lower = **0.66**), BR (middle = **2.0**), and BR (heavy = **3.8**)

The time-weighted average was calculated based on the following equation:

$$TWA = (P_{lower})(BR_{lower}) + (P_{middle})(BR_{middle}) + (P_{high})(BR_{high})$$

A Monte Carlo simulation was run to randomly sample from the probability distribution for ***P***, with each iteration yielding a different estimate of the time-weighted average breathing rate. The summary statistics for the cumulative distribution are given below.

EDF = {percentiles, values} = {0.01, 0.025, 0.05, 0.075, 0.10, 0.25, 0.50, 0.75, 0.90, 0.925, 0.95, 0.975, 0.99}, {0.72, 0.72, 0.72, 0.73, 0.73, 0.80, 1.14, 1.47, 1.96, 2.07, 2.12, 2.45, 2.45}

These data could be incorporated into a probabilistic model directly as an empirical distribution. A beta distribution was fit to the summary statistics because it is both flexible in shape and defined by a minimum and maximum value. The process used to generate the PDF, as described above, will generate a plausible estimate of the minimum (100% of exposure time at lowest breathing rate) and maximum (100% of exposure time at highest breathing rate). This characteristic of the data set lends itself to a close fit to the beta distribution.

Inhalation Rate (IR_air) Office Worker

A deterministic value of 1.1 m^3/hr was used from the 1998 Rocky Flats PPRG spreadsheets.

Inhalation Rate (IR_air) Open Space User

A deterministic value of 1.7 m^3/hr was used from the 1998 Rocky Flats PPRG spreadsheets.

Notes on the Beta Distribution

The following discussion presents basic information on the use and definition of the beta distribution, and summarizes a comparison of the distribution functions used by RESRAD 6.0 and Crystal Ball v. 4.0g. Further information on these distributions can be obtained from the user's manual or help menus included with the respective software.

Why use the Beta Distribution?

The beta distribution is very flexible thanks to its two shape parameters it can assume nearly any shape, including right skewed, left skewed, symmetric, and uniform (rectangular). Most lognormal distributions can be approximated well with a beta distribution. An advantage of the beta distribution is that it is bounded by definition at both a minimum and maximum value. Other distributions may require more arbitrary definitions for truncation limits. This does not mean that use of the beta removes the decision making altogether. As with the lognormal distribution, which is bounded at zero by definition, sometimes a higher "lower limit" is needed. For example, if we describe body weight with a lognormal distribution, it would not make sense to allow for a 0 kg individual, so a truncation limit would be needed to increase the minimum value to a plausible range. The same common sense applications should accompany the use of the beta.

Rescaling and Relocating the Beta Distribution [0, 1]

Most algorithms define the shape of the beta for values in the interval [0, 1]. The distribution can then be rescaled to different units, and relocated, while still maintaining the shape. The algorithms used to accomplish this rescaling and relocating can vary. The easiest and most straightforward approach is to select or fit the two shape parameters for the interval [0, 1] and then adjust the scale as follows:

$$beta_{[\min,\max]} = \min + (\max - \min) \times beta_{[0,1]}$$

Goodness of fit software will fit all four parameters [$\_{1}$, $\_{2}$, min, max] simultaneously. A good test of these parameter estimates would be to rescale a data set so that all values lie within the interval [0, 1] - dividing by the maximum value in the data set is one approach.

The beta distribution as used in RESRAD and Crystal Ball

For the EPA standard risk methodology approach, simply your life by removing the "scaling" parameter in Crystal Ball (i.e., set scaling parameter s = 1.0). Define the assumption cell for the variable as usual, so that it yields a value in the interval [0, 1], then include the min and max in the risk formula as shown above. To convert units of variables defined in the EPA standard risk methodology spreadsheet so that they match the RESRAD units, apply the conversions only to the [min, max]; do not modify the shape parameters. See the Example 1 below for a more visual explanation.

The RESRAD 6.0 Beta Distribution Function

$$f(x) = \frac{(P+Q-1)!(x-Min)^{P-1}(Max-x)^{Q-1}}{(P-1)!(Q-1)!(Max-Min)^{P+Q-1}}$$

where,

P	=	shape parameter (alpha 1 or _1)
Q	=	shape parameter (alpha 2 or _2)
Min	=	minimum
Max	=	maximum

for P > 0 and Q > 0, and Max > Min.

If the generic interval [min, max] is defined as [0, 1] then the equation reduces to

$$f(x) = \frac{(P+Q-1)!(x)^{P-1}(1-x)^{Q-1}}{(P-1)!(Q-1)!}$$

and the beta random variate lies within the interval: 0 < x < 1.

The Crystal Ball Beta Distribution Function

Using the same parameter notation as RESRAD:

$$\mathbf{f(x)} = \frac{\mathbf{(P+Q-1)!(\frac{x}{s})^{P-1}(1-\frac{x}{s})^{Q-1}}}{\mathbf{(P-1)!(Q-1)!}}$$

where

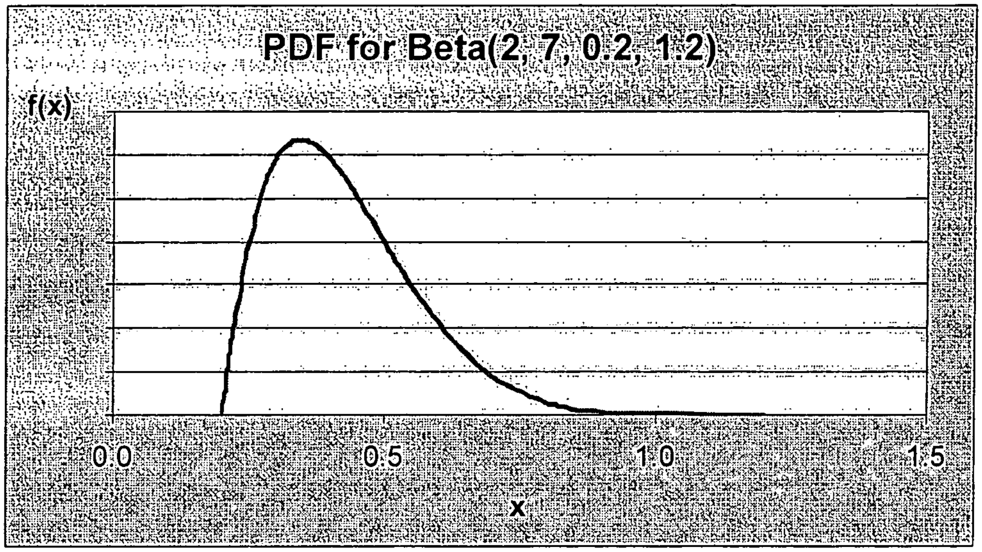
P	=	shape parameter (alpha 1 or _1)
Q	=	shape parameter (alpha 2 or _2)
s	=	scale parameter
Min	=	minimum
Max	=	maximum

for P > 0, Q > 0, (P + Q + 1) < 1750, Max > Min, and s > 0.

This definition will yield a beta random variate that lies within the interval: $0 < x < s$, as well as the interval [min, max]. Since both conditions are satisfied, if the min > 0 or max < s, this can result in a very "truncated" looking distribution. Note that Crystal Ball yields the same equation as RESRAD if (and only if) the scaling factor is set to 1.0.

Example 1. Unit Conversions and the beta distribution, X ~beta(alpha 1, alpha 2).

Assume data are collected for variable X, and fit to a beta distribution: X ~ beta(2,7) with a minimum of 0.2 and maximum of 1.2. Now assume that the units for the variable are converted by multiplying by 10. A new beta distribution is fit to this data set yielding: X ~ beta(2, 7), but with a new minimum of 2.0 and maximum of 12.0 (multiply previous *min* and *max* by 10). Note that the two shape parameters do not change, so the shape of the PDF remains the same in the graphs below. Only the scale of the x-axis is modified by the change in the interval. Parameters are [alpha 1, alpha 2, min, max].



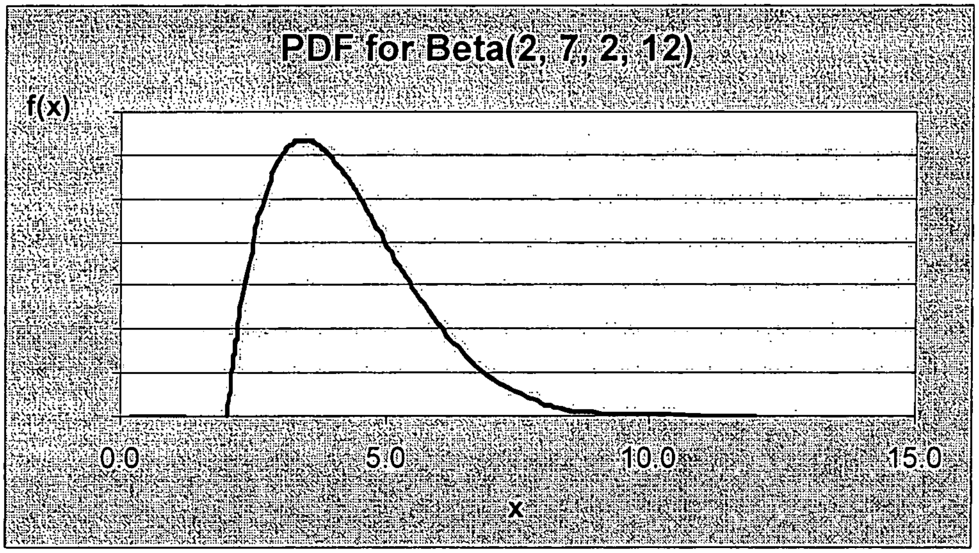


Figure A-11

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Exposure Frequency (EF) Rural Resident

Exposure frequency (EF) refers to the number of days per year that a resident is present at home, rather than at work or on vacation. Given that the toxicity endpoint is a long-term average exposure (the endpoint of concern is cancer), this input variable will represent a long-term average time at the residence. For the rural resident land use scenario, it is assumed that if an individual is at home, they may be exposed via one or more exposure pathways for 24 hours per day (see Exposure Time). For this analysis, no distinction is made between exposure frequencies for men and women, or for children and adults. The maximum number of days per year is 365 days.

The U.S. EPA Exposure Factors Handbook (EPA, 1998) summarizes survey data on population mobility for the U.S. population. The sample sizes for the major studies are very large (n > 1000), reflecting national surveys. The difficulty in estimating population activity patterns and mobility from a survey is that it represents a snapshot in time, and there is uncertainty in determining the total duration that an individual will reside at the same house (see Exposure Duration). Extrapolations to a long time periods are required since personal diaries cover short periods of time. However, there is less uncertainty associated with estimating the days per year that an individual spends time at home.

The Superfund default central tendency estimate for residential exposure frequency is 234 days/year, which corresponds to the fraction of time spent at home (64%) for both men and women based on a study of time use patterns summarized in 1990. In other words, the available data suggest that, on average, individuals spend approximately two-thirds of the year at home.

Probability Distribution

For this analysis, a probability distribution was generated from the central tendency estimate given by U.S. EPA exposure factors handbook (234 days/year) and professional judgment regarding a plausible range among a residential population. The maximum value of 350 days was selected to reflect an average of approximately two weeks per year spent away from home, either on family vacation or business travel. A minimum of 175 days/year was selected to reflect a minimum of approximately 50% of the year spent at home.

Given reliable information regarding the central tendency, and plausible estimate for the minimum and maximum, the following triangular distribution was selected to represent variability in exposure frequency among rural residential populations:

EF ~ Triangular (175, 234, 350) days/year

The parameters for the triangular distribution are as follows:

- minimum 175 days/year
- mode 234 days/year
- minimum 350 days/year

The mode characterizes the "most likely" value and will equal the mean for distributions that are symmetrical. Figure A-12 presents the probability density and cumulative distribution views for this distributions. The mean, 90th, 95th and 99th percentiles are 253, 305, 318, and 336 days/year.

Uncertainties in the Probability Distribution

The triangular distribution is a reasonable approximation for the "true" distribution for exposure frequency given that the variable is truncated at the high end by definition (i.e., 365 days per year). It may be possible to obtain the original survey data results that formed the basis for the central tendency estimate (CTE) recommended by EPA for use in Superfund risk assessments. However, it is expected that use of an alternative right-skewed (and truncated) distribution would yield very similar percentile estimates, and would therefore have only a minor effect on the risk estimates.

Use of 350 days/year as a high-end truncation limit is viewed as a reasonably conservative estimate of exposure frequency in the absence of site-specific data.

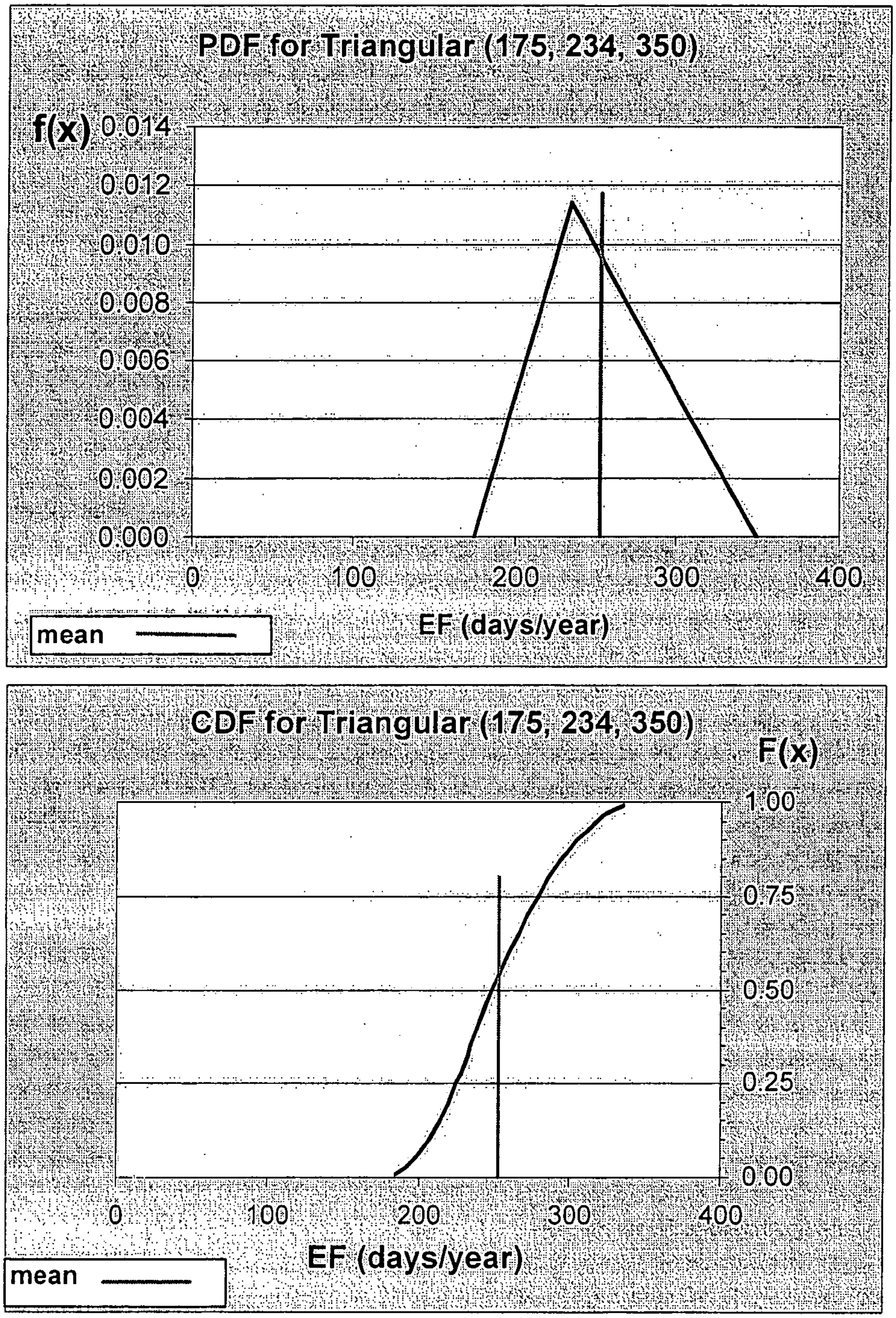


Figure A-12. Probability density function (PDF) and cumulative distribution function (CDF) views of the triangular distribution for exposure frequency (days/year) for the rural resident

Exposure Frequency (EF) Wildlife Refuge Worker

For the wildlife refuge worker scenario, exposure frequency represents the average number of days per year that a refuge worker spends on site. National survey data on occupational activity patterns are maintained by the Bureau of Labor Statistics. The Superfund default central tendency and reasonable maximum exposure estimates for both full time and part-time workers is 219 days/year and 250 days/year, respectively. The 250 days/year reflects an individual who works 5 days per week for 50 weeks of the year (thereby taking a single 2-week vacation, for example). These estimates are based on national survey data of the U.S. population from 1991.

Since it is likely that different occupations may reflect substantially different activity patterns, ideally a sub-category representative of wildlife refuge workers would be used to estimate exposure frequency. Such occupation-specific information has been obtained by the U.S. Fish and Wildlife Service in a National Wildlife Refuge Survey, in which wildlife refuge workers were interviewed from three refuges (Crab Orchard, IL; Malheur, OR; and Minnesota Valley, MN). Data for 33 wildlife refuge workers are summarized in the RMA (1994). The responses allow for estimates of either hours per day or days per year. While the sample size is relatively small, the estimates are similar to that of the national survey data, and provide a more occupation-specific data set for the exposure scenario characterized in this analysis.

Probability Distribution

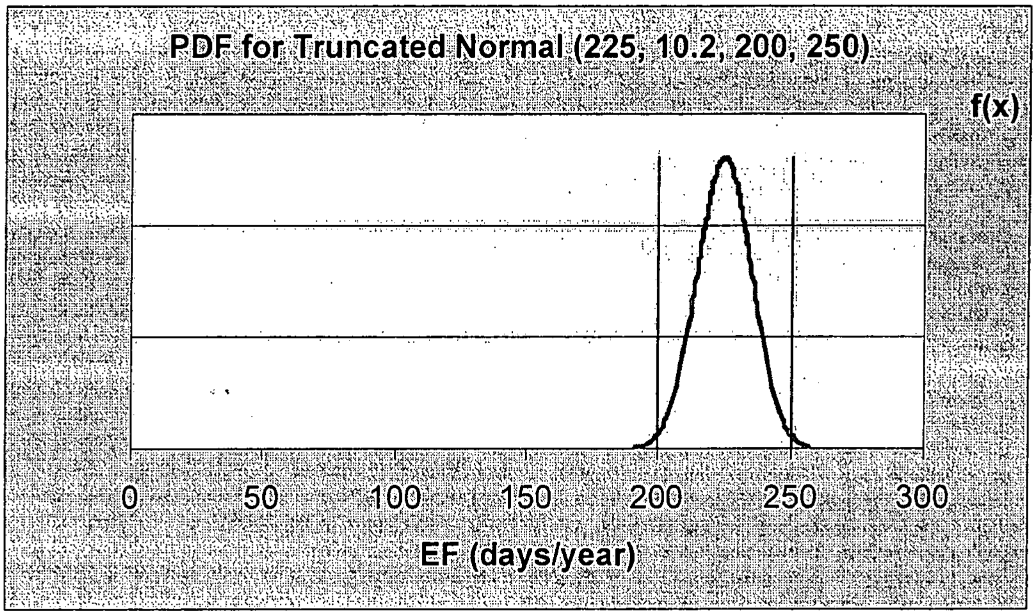
The following probability distribution is recommended for use in risk equations that are based on U.S. EPA Risk Assessment Guidance for Superfund (EPA STANDARD RISK METHODOLOGY) in order to characterize *interindividual* variability in exposure frequency among wildlife refuge workers:

EF ~ Truncated Normal (225, 10.23, 200, 250) days/year

The truncated normal distribution is defined by four parameters:

• arithmetic mean	225	days/year
• standard deviation	10.23	days/year
• minimum	200	days/year
• maximum	250	days/year

The probability distribution (PDF and CDF) is shown in Figure A2. Given that a normal distribution has infinite lower and upper tails, it is reasonable to truncate the distribution at plausible bounds. The affect of the truncation limit is to alter the original parameter estimates (mean, standard deviation) that is effectively used in a Monte Carlo simulation. For this analysis, the coefficient of variation (CV = stdev / mean) is very low (0.05), so truncating at 200 and 250 days/year has a minimal effect. These truncation limits remove 0.7% of the tail at both ends, and due to the symmetrical shape, there is no change in the mean or standard deviation.



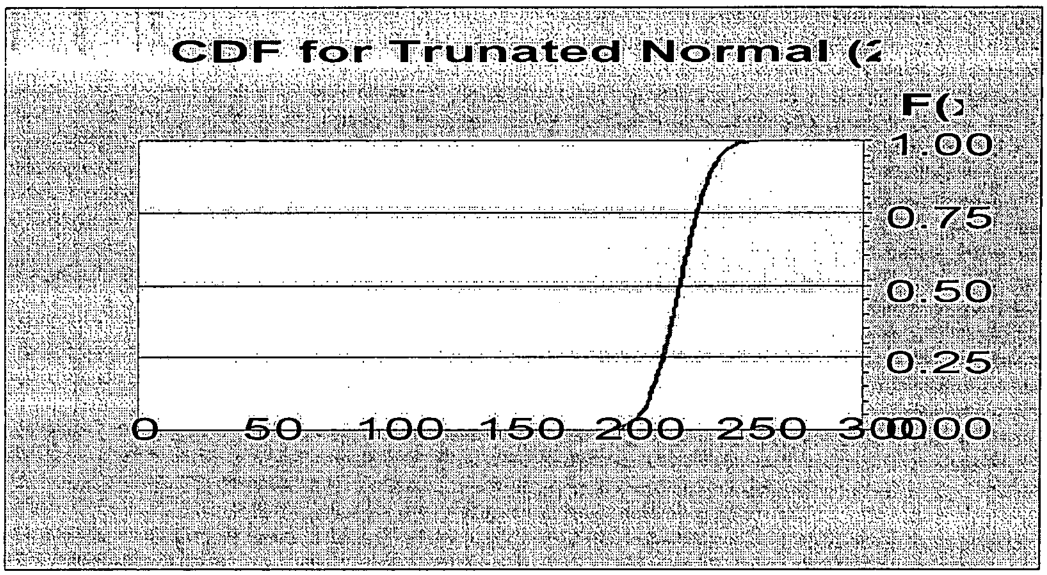


Figure A-13 Probability density function (PDF) and cumulative distribution function (CDF) views of the truncated normal distribution for (adult) exposure frequency (days/year) for the wildlife refuge worker.

Uncertainties in the Probability Distribution

The use of a normal distribution is supported by the data reported by U.S. Fish and Wildlife on wildlife refuge workers in three different locations. The arithmetic mean (225 days/year) is slightly greater than the central tendency estimate reported by the Bureau of Labor Statistics for all occupations (219 days/year). The maximum value of 250 days/year is consistent with the RME estimate recommended for use at Superfund sites, and may be viewed as a reasonable upper bound for individuals who work week days only, and take two weeks of vacation per year. The lower bound of 200 days per year suggests that the range among different workers at the refuge is relatively narrow (i.e., 50 days).

Exposure Frequency (EF) Office Worker

Deterministic value of 250 days/yr used in 1998 Rocky Flats PPRG spreadsheets.

Exposure Frequency (EF) Open Space User

Deterministic value of 100 days/yr used in 1998 Rocky Flats PPRG spreadsheets.

Exposure Duration (ED) Rural Resident

Exposure duration (ED) refers to the number of years that a resident is present at the same residence. For the rural resident land use scenario, both children and adults comprise the population of concern, and exposure is assumed to begin at birth. Census data provide representations of a cross-section of the population at specific points in time, but the surveys are not designed to follow individual families through time (U.S. EPA, 1998). The U.S. EPA Exposure Factors Handbook (EPA, 1997) summarizes the key studies on population mobility. These studies use a variety of methods to estimate residential tenures, including, 1) calculate the average current and total residence times; 2) model current residence time; and 3) estimate the residential occupancy period. Each of the key studies and methodologies provides similar estimates as summarized in Table A-13.

Table A-13. Summary of Key Studies for Residential Exposure Duration, based on U.S. EPA (1998), Table 15-174.

Study	Summary Statistics (years)	Methodology
Isreali and Nelson, 1992	mean = 4.6 1/6 of a lifetime of 70 years, or 11.7 years	average current and total residence times
US Bureau of the Census, 1993	50^{th} percentile = 9 years 90^{th} percentile = 33 years	current residence time
Johnson and Capel, 1992	mean = 12 years 90^{th} percentile = 26 years 95^{th} percentile = 33 years 99^{th} percentile = 47 years	residential occupancy period

Probability Distribution

For this analysis, a probability distribution was generated from the empirical distribution function reported by Johnson and Capel (1992) for n = 500,000 simulated individuals (both male and female) given in Table A-14.

Table A-14. Empirical cumulative distribution function for residential occupancy period reported by Johnson and Capel (1992), based on U.S. EPA (1998), Table 15-167.

Percentile*	**Years**	**Percentile**	**Years**
0.05	2	0.95	33
0.10	2	0.98	41
0.25	3	0.99	47
0.50	9	0.995	51
0.75	16	0.998	55
0.90	26	0.999	59

* the maximum observed value was 87 years.

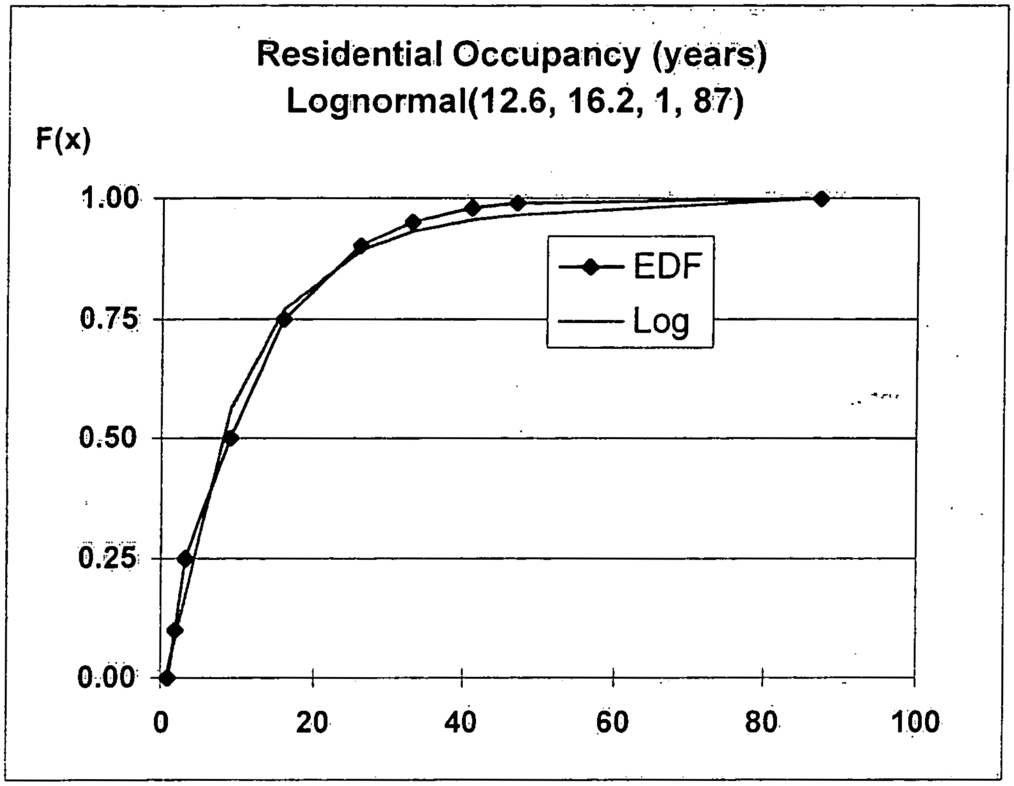


Figure A-14 Comparison of ECDF and truncated lognormal distribution for residential occupancy period (ED, years).

These data were fit to a lognormal distribution using least squares regression to estimate the arithmetic mean of 12.6 years and standard deviation of 16.2 years. A comparison of the EDF to the fitted lognormal distribution is given by Figure A1. Truncation limits of 1 and 87 are based on professional judgment that the maximum observed values are plausible bounds given the large sample size of the survey. The corresponding probability distribution function is shown in Figure A-15.

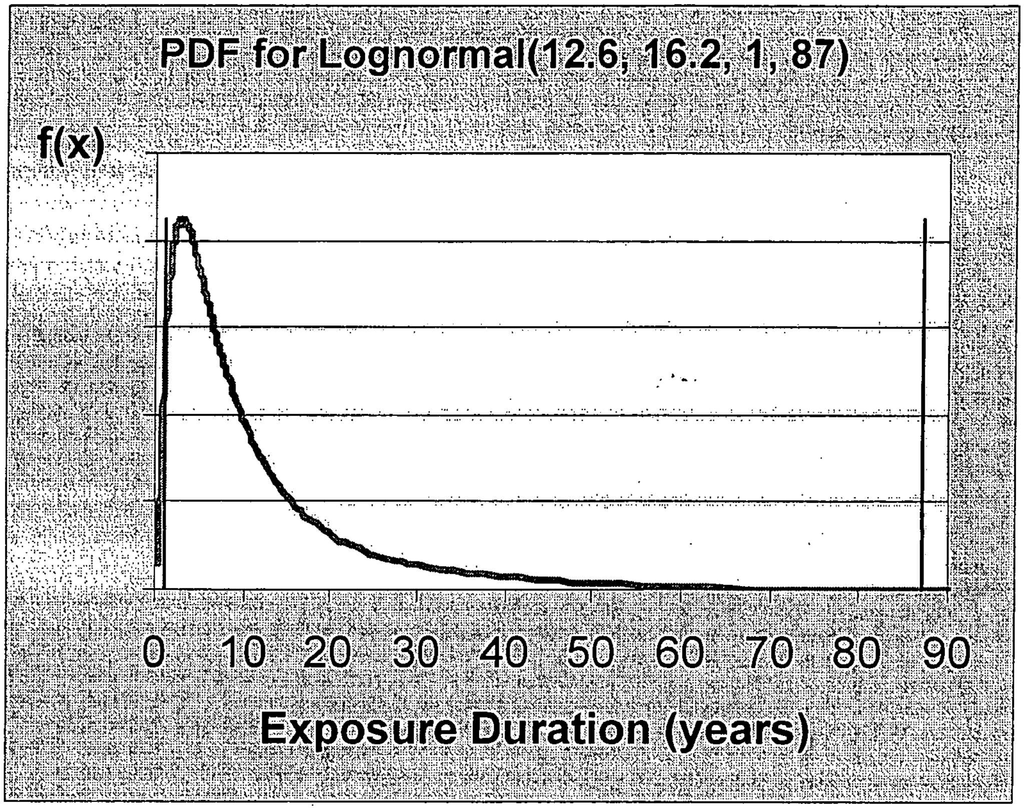


Figure A-15. Probability density function (PDF) and cumulative distribution function (CDF) views of the lognormal distribution for exposure duration (years) for the rural resident.

Given reliable fit to the empirical distribution function the following lognormal distribution was selected to represent variability in exposure duration among rural residential populations:

ED ~ Truncated Lognormal (12.6, 16.2, 1, 87) years

The parameters for the truncated lognormal distribution are as follows:

• arithmetic mean	12.6	years
• arithmetic standard deviation	16.2	years
• minimum	1	year
• maximum	87	years

This use of truncation limits on this distribution does have a moderate effect on the parameter estimates used in the Monte Carlo simulation. The maximum value of 87 years truncates the distribution at the 99.3rd percentile, while the minimum value truncates the distribution at the 1.9th percentile. These truncation limits have the combined effect of reducing the mean to 12.0 years (4.8%) and reducing the standard deviation to 12.3 years (24.1%). This change reflects the relative high coefficient of variation for this distribution (CV = stdev/mean = 1.3), however, the maximum of 87 years is considered to be a reasonable approximation of an individual who lives at the same residence their entire life.

The 50th, 90th, 95th and 99th percentiles of this distribution are 7.7, 27.4, 39.3, and 77.0 years.

Uncertainties in the Probability Distribution

The is relatively high confidence in the data set and probability distribution used to characterize variability in residential exposure duration. The standard RME point estimate for use in Superfund risk assessments (for cancer) is 30 years, which is approximately the 91st percentile of this distribution.

Exposure Duration (ED) Wildlife Refuge Worker

For the wildlife refuge worker scenario, exposure duration represents the number of years that a refuge worker spends on site. National survey data on occupational activity patterns are maintained by the U.S. Bureau of Labor Statistics. The Superfund default reasonable maximum exposure estimate for both full time and part-time workers is 25 years, based on the 95th percentile of the number of years worked at the same location reported in 1990.

There are a wide range of reported job tenures among different categories of occupations. The U.S. EPA Exposure Factors Handbook (1998, Table 15A-7) summarizes data reported by Carey (1988) for 109 million adults (16+ years). The median job tenure for the entire survey (all ages, male and female) is 6.6 years, however this varies by occupation and age. Examples of some of median job tenure for selected occupations are given in Table A-15.

Table A-15 Median job tenure for selected occupations based on Carey (1988) as reported by U.S. EPA (1988), Table 15A-7.

Occupation	Median Tenure (yrs)	Occupation	Median Tenure (yrs)
Barbers	24.8	Health Technologists and Technicians	6.3
Farmers, except horticulture	21.1	Supervisors; Ag Operations	5.2
Construction Inspectors	10.7	Machine Operators	4.5
Administrators and Officials, Public Admin	8.9	Biological Technicians	4.4
Surveying and Mapping Technicians	8.6	Animal Caretakers, except farm	3.5
Science Technicians	7.0	Information Clerks	2.7

The major limitation in using these data to estimate ED for risk assessment is that they reflect time spent in an occupation rather than time spent at a particular job site. In addition, these data reflect median job tenures, and whereas the complete distribution of tenures within a category are of interest. Ideally a sub-category representative of wildlife refuge workers at one site would be used to estimate exposure duration. Such occupation-specific information has been obtained by the U.S. Fish and Wildlife Service in a National Wildlife Refuge Survey, in which wildlife refuge workers were interviewed from three refuges (Crab Orchard, IL; Malheur, OR; and Minnesota Valley, MN). Data for 80 wildlife refuge workers are summarized in the RMA (1994). Of these workers, 33 values reflect incomplete tenures, and 47 values reflect completed tenures. The responses allow for estimates of years spent at one refuge, regardless of whether job activities changed. While the sample size is relatively small, the estimates are similar to that of the national survey data, and provide a more occupation-specific data set for the exposure scenario characterized in this analysis.

Probability Distribution

The following probability distribution is recommended for use in risk equations that are based on U.S. EPA Risk Assessment Guidance for Superfund (EPA Standard Risk Methodology) in order to characterize *interindividual* variability in exposure duration among wildlife refuge workers:

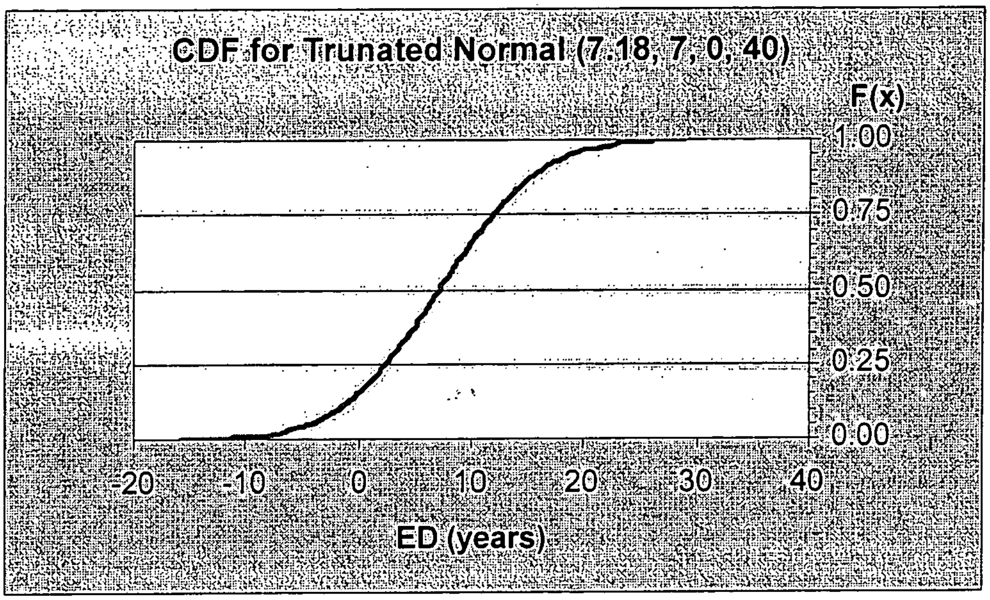
ED ~ Truncated Normal (7.18, 7, 0, 40) years

The truncated normal distribution is defined by four parameters:

- arithmetic mean 7.18 years
- arithmetic standard deviation 7 years
- minimum 0 years
- maximum 40 years

The probability distribution (PDF and CDF) is shown in Figure A3. Given that a normal distribution has infinite lower upper tails, it is reasonable to truncate the distribution at a plausible bounds. A minimum of 0 was chosen to avoid negative values, and a maximum of 40 years was chosen to be approximately 5 standard deviations from the mean, so as to minimize the effect on the parameter estimates in the Monte Carlo simulation. The effect of the truncation limit is to alter the original parameter estimates (mean, standard deviation) to (9.1, 5.6), an increase of 27% in the mean and reduction of 27% in the standard deviation. It is clear from Figure A3 that the truncation limit reduces a significant fraction of the low-end values; in such cases, it is generally preferable to use an alternative distribution that requires less truncation (e.g., lognormal). This was not done for this analysis given that the data were not reported in a manner that would allow for exploration of alternative PDFs.

The 50^{th}, 90^{th}, 95^{th}, and 99^{th} percentiles of this distribution are 7.2, 16.2, 18.7, and 23.5 years, respectively.



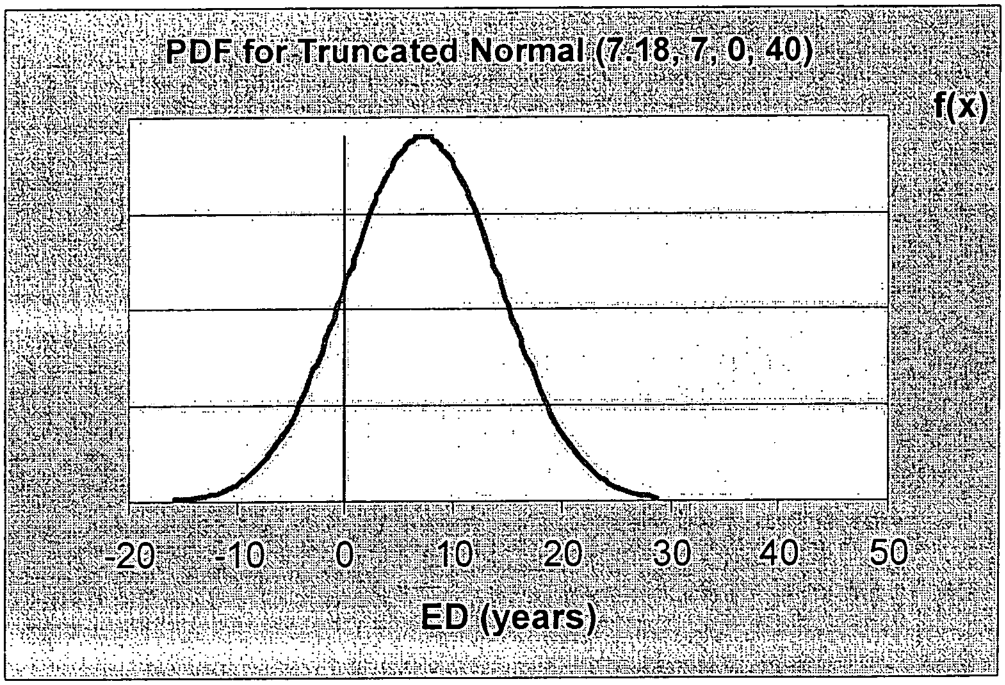


Figure A-16 Probability density function (PDF) and cumulative distribution function (CDF) views of the truncated normal distribution for exposure duration (years) for the wildlife refuge worker.

Uncertainties in the Probability Distribution

The use of a truncated normal distribution is supported by the data reported by U.S. Fish and Wildlife on wildlife refuge workers in three different locations. Data from Carey et al. (1988) for the U.S. population suggest that the highest median tenure at one job is less than 30 years, and the median tenure of all occupations is 6.6 years. The tenure for biological technicians is reported to be 4.4. years. The use of a normal distribution is professional judgment given the reported arithmetic mean and standard deviation for n = 33 biological refuge workers (or 80 tenures). The U.S. Fish and Wildlife Service fit the normal distribution to these data, although an alternative bounded distribution (e.g., beta, lognormal) may be preferable given the significant fraction of low-end values that are truncated below 0.

Exposure Duration (ED) Office Worker

Deterministic value of 25 years used in 1998 Rocky Flats PPRG spreadsheets.

Exposure Duration (ED) Open Space User

Deterministic value of 30 years used in 1998 Rocky Flats PPRG spreadsheets.

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APPENDIX B - DESCRIPTION OF EPA'S RISK ASSESSMENT EQUATIONS AND PARAMETER VALUES

The following summary gives the risk equations by exposure pathway that were used to calculate risk given a PRG. In the Excel spreadsheets used to calculate PRSALs, the same equations were applied by rearranging the equations to solve for PRG. Following the equations is a summary sheet that gives the point estimates and probability distributions used in these equations.

A. Risk Equations for Residential Scenario

Receptor Population:	combined child (1 - 6 yrs) and adult (7+ yrs)
Health Endpoint:	cancer (chronic exposure)
Exposure Pathways:	inhalation, soil ingestion, home-grown diet, external exposure

Inhalation Pathway

$$Risk_{inhalation} = PRG \times IR_{a\_age} \times ED \times EF \times \frac{ET}{24} \times ML \times CF_1 \times [ET_0 + ET_i \times DF_i] \times SF_{inh}$$

where,

$Risk_{inhalation}$	= excess lifetime cancer risk from inhalation of radionuclide
PRG	= preliminary remediation goal; concentration in soil (pCi/g)
IR_{a_age}	= age-adjusted inhalation rate (m^3/day) (see below)
ED	= exposure duration for chronic exposure (yr)
EF	= exposure frequency (day/yr)
ET	= exposure time at residence (hrs/day) [divided by 24 hrs/day]
ML	= mass loading (_g/m^3)
CF_1	= conversion factor (10^{-6} g/_g)
ET_o	= exposure time fraction, outdoors (unit less)
ET_i	= exposure time fraction, indoors (unit less)
Dfi	= dilution factor for indoor inhalation (unit less)
SF_{inh}	= inhalation slope factor (risk/pCi)

$$IR_{a\_age} = \frac{(IR_{a\_child} \times ED_{child}) + (IR_{a\_adult} \times ED_{adult})}{ED}$$

where,

IR_{a_child}	= inhalation rate for children (m^3/day)
IR_{a_adult}	= inhalation rate for adults (m^3/day)
ED_{child}	= exposure duration during childhood (yr)
ED_{adult}	= exposure duration during adulthood (yr)

Residential Scenario (cont'd)

Soil Ingestion Pathway

$$Risk_{soil} = PRG \leftarrow IR_{a\_age} \leftarrow ED \leftarrow EF \leftarrow CF_2 \leftarrow SF_{soil}$$

where

$Risk_{soil}$	=	excess lifetime cancer risk from ingestion of radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
*IR_{s_age}	=	age-adjusted soil ingestion rate (mg/day)
ED	=	exposure duration (yr)
EF	=	exposure frequency (day/yr)
CF_2	=	conversion factor (10^{-3} g/mg)
SF_{soil}	=	oral slope factor (risk/pCi)

*Note that ingestion rates are age-specific, so each ingestion rate is estimated for both children and adults, and weighted based on exposure duration:

$$IR_{s\_age} = \frac{\left(IR_{s\_child} \leftarrow ED_{child}\right) + \left(IR_{s\_adult} \leftarrow ED_{adult}\right)}{ED}$$

where,

IR_{s_child}	=	inhalation rate for children (mg/day)
IR_{s_adult}	=	inhalation rate for adults (mg/day)
ED_{child}	=	exposure duration during childhood (yr)
ED_{adult}	=	exposure duration during adulthood (yr)

Food Ingestion Pathway

$$Risk_{food} = \left(C_{pv} + C_{pr} + C_{pd}\right) \leftarrow CR_{food} \leftarrow ED \leftarrow SF_0$$

where,

$Risk_{food}$	=	excess lifetime cancer risk from ingestion of radionuclide in home-grown fruit, vegetables, and grain
C_{pv}	=	concentration in plant, vegetative fraction (pCi/kg)
C_{pr}	=	concentration in plant, root fraction (pCi/kg)
C_{pd}	=	concentration on plant, deposition fraction (pCi/kg)
CR_{food}	=	consumption rate of homegrown fruit, vegetables, and grain (kg/yr)
ED	=	exposure duration for combined child and adult (yr)
SF_o	=	oral slope factor (risk/pCi)

Residential Scenario (cont'd)

$$C_{pv} = PRG \times CF_1 \times B_v \times DWC_v \times F_v$$

where,

PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
CF_1	=	conversion factor (10^3 g/kg)
B_v	=	soil-plant conversion factor, vegetation (unit less)
DWC_v	=	dry weight conversion factor, vegetative (pCi/kg)
F_v	=	fraction of total vegetable intake from vegetative portion (unit less)

$$C_{pr} = PRG \times CF_1 \times B_r \times DWC_r \times F_r$$

where,

PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
CF_1	=	conversion factor (10^3 g/kg)
B_r	=	soil-plant conversion factor, roots (unit less)
DWC_r	=	dry weight conversion factor, roots (pCi/kg)
F_r	=	fraction of total vegetable intake from root portion ($F_r = 1 - F_v$) (unit less)

$$C_{pd} = PRG \times ML_p \times LT$$

where,

PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
MLp	=	mass loading factor for plant surfaces (g/m^3)
LT	=	lumping term for deposition (m^3/kg)

$$CR_{food} = CR_{veg} + CR_{fruit} + \left(CR_{grain} \times HG_{grain}\right)$$

where,

$*CR_{food}$	=	consumption rate of homegrown vegetables, fruit, and grain (kg/yr)
CR_{veg}	=	consumption rate of homegrown vegetables (kg/yr)
CR_{fruit}	=	consumption rate of homegrown fruit (kg/yr)
CR_{grain}	=	consumption rate of total grain (kg/yr)
HG_{grain}	=	homegrown fraction for grain (unit less)

** Note that ingestion rates are age-specific, so each consumption rate is estimated for both children and adults, and weighted based on exposure duration, as given by the following equation.*

Residential Scenario (cont'd)

$$CR_{i\_age} = \frac{\left(CR_{i\_child} \times ED_{child}\right) + \left(CR_{i\_adul} \times ED_{adult}\right)}{ED}$$

where,

CR_{i_age}	=	age-adjusted consumption rate of i^{th} food type (kg/yr)
CR_{i_child}	=	consumption rate of i^{th} food type for children (kg/yr)
CR_{i_adult}	=	consumption rate of i^{th} food type for adults (kg/yr)
ED_{child}	=	exposure duration during childhood (yr)
ED_{adult}	=	exposure duration during adulthood (yr)

External Exposure Pathway[2]

$$Risk_{ext} = PRG \times ACF \times \left(\frac{EF}{365}\right) \times ED \times \left[ET_o + ET_i \times (1 - S_e)\right] \times SF_{ext}$$

where,

$Risk_{ext}$	=	excess lifetime cancer risk from direct external exposure to radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
ACF	=	area correction factor (unit less)
EF	=	exposure frequency (day/yr)
ED_e	=	exposure duration (yr)
ET_o	=	exposure time fraction, outdoor (unit less)
ET_i	=	exposure time fraction, indoor (unit less)
S_e	=	gamma shielding factor (unit less)
SF_{ext}	=	oral slope factor (risk/yr per pCi/g)

[2] Eq. 4 of U.S. EPA. 2000. Soil Screening Guidance for Radionuclides: User's Guide. EPA/540-R-00-007.

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B. Risk Equations for Occupational Scenario (Office, Wildlife Refuge)

Receptor Population: adult (18+ yrs)
Health Endpoint: cancer (chronic exposure)
Exposure Pathways: inhalation, soil ingestion, external exposure

Inhalation Pathway

$$Risk_{inhalation} = PRG \leftrightarrow IR \leftrightarrow ED \leftrightarrow EF \leftrightarrow ET \leftrightarrow ML \leftrightarrow CF_1 \leftrightarrow [ET_o + ET_i \leftrightarrow DF_i] \leftrightarrow SF_{inh}$$

where,

$Risk_{inhalation}$	=	excess lifetime cancer risk from inhalation of radionuclide
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
IR	=	inhalation rate (m^3/hr)
ED	=	exposure duration for chronic exposure (yr)
EF	=	exposure frequency (day/yr)
ET	=	exposure time at workplace (hrs/day)
ML	=	mass loading (_g/m^3)
CF_1	=	conversion factor (10^{-6} g/_g)
ET_o	=	exposure time fraction, outdoors (unit less)
ET_i	=	exposure time fraction, indoors (unit less)
DFi	=	dilution factor for indoor inhalation (unit less)
SF_{inh}	=	inhalation slope factor (risk/pCi)

Soil Ingestion Pathway

$$Risk_{soil} = PRG \leftrightarrow IR_s \leftrightarrow ED \leftrightarrow EF \leftrightarrow CF_2 \leftrightarrow SF_{soil}$$

where,

$Risk_{soil}$	=	excess lifetime cancer risk from ingestion of radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
IR_s	=	adult soil ingestion rate (mg/day)
ED	=	exposure duration (yr)
EF	=	exposure frequency (day/yr)
CF_2	=	conversion factor (10^{-3} g/mg)
SF_{soil}	=	oral slope factor (risk/pCi)

Occupational Scenario (Office, Wildlife Refuge)

External Exposure Pathway

$$Risk_{ext} = PRG \times ACF \times \left(\frac{EF}{365}\right) \times ED \times \left[ET_o + ET_i \times (1 - S_e)\right] \times SF_{ext}$$

where,

$Risk_{ext}$	=	excess lifetime cancer risk from direct external exposure to radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
ACF	=	area correction factor (unit less)
EF	=	exposure frequency (day/yr)
ED	=	exposure duration (yr)
ET_o	=	exposure time fraction, outdoor (unit less)
ET_i	=	exposure time fraction, indoor (unit less)
S_e	=	gamma shielding factor (unit less)
SF_{ext}	=	oral slope factor (risk/yr per pCi/g)

C. Risk Equations for Open Space User

Receptor Population: combined child (1 - 6 yrs) and adult (7+ yrs)
Health Endpoint: cancer (chronic exposure)
Exposure Pathways: inhalation, soil ingestion, external exposure

Inhalation Pathway

$$Risk_{inhalation} = PRG \times IR_{a\_age} \times ED \times EF \times ET \times ML \times CF_1 \times SF_{inh}$$

where,

$Risk_{inhalation}$	=	excess lifetime cancer risk from inhalation of radionuclide
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
IR_{a_age}	=	age-adjusted inhalation rate (m^3/day) (see below)
ED	=	exposure duration for chronic exposure (yr)
EF	=	exposure frequency (day/yr)
ET	=	exposure time at open space (hrs/day)
ML	=	mass loading (_g/m^3)
CF_1	=	conversion factor (10^{-6} g/_g)
SF_{inh}	=	inhalation slope factor (risk/pCi)

$$IR_{a\_age} = \frac{(IR_{a\_child} \times ED_{child}) + (IR_{a\_adult} \times ED_{adult})}{ED}$$

where,

IR_{a_child}	=	inhalation rate for children (m^3/day)
IR_{a_adult}	=	inhalation rate for adults (m^3/day)
ED_{child}	=	exposure duration during childhood (yr)
ED_{adult}	=	exposure duration during adulthood (yr)

Open Space User (cont'd)

Soil Ingestion Pathway

$$Risk_{soil} = PRG \times IR_{s\_age} \times ED \times EF \times CF_2 \times SF_{soil}$$

where

$Risk_{soil}$	=	excess lifetime cancer risk from ingestion of radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
*IR_{s_age}	=	age-adjusted soil ingestion rate (mg/day)
ED	=	exposure duration (yr)
EF	=	exposure frequency (day/yr)
CF_2	=	conversion factor (10^{-3} g/mg)
SF_{soil}	=	oral slope factor (risk/pCi)

*Note that ingestion rates are age-specific, so each ingestion rate is estimated for both children and adults, and weighted based on exposure duration:

$$IR_{s\_age} = \frac{(IR_{s\_child} \times ED_{child}) + (IR_{s\_adult} \times ED_{adult})}{ED}$$

where,

IR_{s_child}	=	inhalation rate for children (mg/day)
IR_{s_adult}	=	inhalation rate for adults (mg/day)
ED_{child}	=	exposure duration during childhood (yr)
ED_{adult}	=	exposure duration during adulthood (yr)

External Exposure Pathway

$$Risk_{ext} = PRG \times ACF \times \left(\frac{EF}{365}\right) \times ED \times [ET_o + ET_i \times (1 - S_e)] \times SF_{ext}$$

where,

$Risk_{ext}$	=	excess lifetime cancer risk from external exposure to radionuclide in soil
PRG	=	preliminary remediation goal; concentration in soil (pCi/g)
ACF	=	area correction factor (unit less)
EF	=	exposure frequency (day/yr)
ED_e	=	exposure duration (yr)
ET_o	=	exposure time fraction, outdoor (unit less)
ET_i	=	exposure time fraction, indoor (unit less)
S_e	=	gamma shielding factor (unit less)
SF_{ext}	=	oral slope factor (risk/yr per pCi/g)

Appendix C - Risk Based Spreadsheets and Instructions for Use for Probabilistic Calculations

This appendix describes the Excel spreadsheets that were developed to obtain both point estimates (i.e., deterministic) and probabilistic estimates of risk and/or risk-based soil action levels (RSALs). In addition, instructions are provided on how to use Crystal Ball, the add-in software to Excel needed to execute the Monte Carlo simulations and reproduce the results presented in the main report. Appendix B presents a detailed description of the equations that were used to calculate risk given a soil concentration of each radionuclide. These same equations were applied to calculate RSALs (by rearranging the equation to calculate RSAL given a target risk level). Appendix A presents a detailed description of the derivation of probability distributions and parameter values for exposure variables identified by the sensitivity analysis as important sources of variability or uncertainty.

A. Excel Spreadsheets

Table C-1 lists the spreadsheets that were developed for calculating point estimates and probabilistic estimates of risk and RSAL. A separate spreadsheet is available for each of the four exposure scenarios: 1) residential rancher; 2) wildlife refuge worker; 3) office worker; 4) open space user.

Table C-1. Excel spreadsheets developed for calculating risks and RSALs with EPA standard risk methodology equations.

Excel Spreadsheet	Exposure Scenario	Exposure Pathways			
		Inhalation	Soil	Food	External
EPA STANDARD RISK METHODOLOGY_resident.xls	Rural Resident	X	X	X	X
EPA STANDARD RISK METHODOLOGY_wildlife.xls	Wildlife Refuge Worker	X	X		X
EPA STANDARD RISK METHODOLOGY_office.xls	Office Worker	X	X		X
EPA STANDARD RISK METHODOLOGY_open.xls	Open Space User	X	X		X

The following features are available on each spreadsheet:

1. Calculate either **risk** or **RSAL** for each of the 5 radionuclides (i.e., Am-241, Pu-239, Ur-234, Ur-235, Ur-238). The spreadsheet automatically sums risks across exposure pathways (see Table C-1), and calculates the percent contribution of each pathway. Note that RSALs are called Preliminary Remediation Goals (PPRGs) in the spreadsheets.

2. Select **point estimates** or **probability distributions** for input variables in the equations by using the toggle provided at the top of the spreadsheet (see Figure C-1). It is important that the toggle be set to probabilistic estimates prior to running a Monte Carlo simulation. Instructions for running Monte Carlo simulations with Crystal Ball are given below.

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3. Calculate the **percent contribution of each exposure pathway** for each radionuclide. If the spreadsheet is used to calculate risk, the user must specify a concentration (pCi/g) for the radionuclides (i.e., cell **C3**). This concentration is applied to each radionuclide. If the spreadsheet is used to calculate RSAL, the user must specify the Target Risk level (e.g. 1E-04, 1E-05, 1E-06) using cell **J4**. This target risk is applied to each radionuclide. Two observations should be noted about these summary statistics:

   a. Because the percent contribution by pathway is independent of the chemical concentration that is selected, the results given in cells **N6 : R10** apply to both the risk and RSAL calculations. For example, using the Point Estimate setting, and a soil concentration for Am-241 of 100 pCi/g, the total risk is 1.4E-04, and the percent contribution of the soil ingestion pathway is 19.0 percent. If the soil concentration is doubled to 200 pCi/g, the total risk doubles to 2.9E-04, but the percent contribution of the soil pathway remains at 19.0 .

   b. When the point estimate option is selected, there will always be only 1 set of results for a given choice of soil concentration or target risk. However, when a probabilistic estimate is selected, the spreadsheets will display one set of random values for results. This means that every time the spreadsheet is reopened, a different set of values will be seen for the following***: risk results (cells **C6 : G10**), input variables (column **F**), percent contribution to risk (cells **N6: R10**). In order to obtain summary statistics for the probabilistic approach, the user needs to run a Monte Carlo simulation using Crystal Ball.

> ***NOTE: Crystal Ball requires a "place-holder cell" be set aside for each input variable. Cells under the heading "Probability Distribution, Value" in **column F** have been designated as the "place holder cells". This particular set of cells allows the computer program to select values from probability distributions while running a Monte Carlo simulation. The values in these cells should be considered random, and should NOT be interpreted as having any correspondence with the point estimates that have been defined for the input variables. See the warning note on each worksheet, as shown in Figure C-1.

4. Comment fields have been extensively used in each spreadsheet to provide additional explanations to the user. Cells with comment fields are denoted by the red triangle in the upper right corner. For example, in the EPA STANDARD RISK METHODOLOGY_resident.xls spreadsheet, the following comment is attached to cell **D16** to explain the units for inhalation rate: *average daily inhalation rate given as m^3/24hr because it may be modified by exposure time (ET).*

5. The slope factors are provided in a separate tab in each spreadsheet called "toxicity". Several different references were evaluated to determine the appropriate slope.

Point Estimates for Probabilistic Estimates

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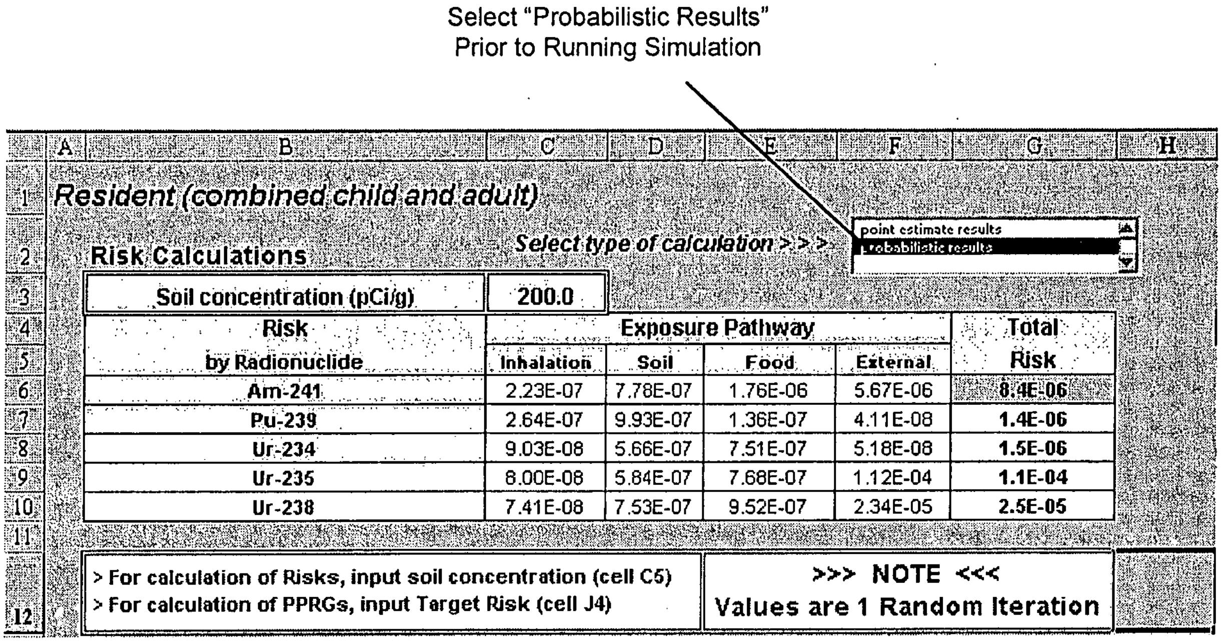
Instructions are provided at the beginning of each Excel spreadsheet to explain the steps in calculating point estimates or probabilistic estimates of risk or RSALs. Table C-2 gives an example of the instructions for the Rural Resident scenario. The following discussion provides the same information in more detail.

Each spreadsheet can be used to calculate risk or RSAL using either point estimates or probability distributions. A toggle is provided at the top of each spreadsheet, as shown in Figure C-1. It is important that this toggle be set to "probabilistic estimates" prior to running a Monte Carlo simulation.

Figure C-1. Toggle to select between Point Estimate Results and Probabilistic Results for the Rural Resident scenario. This is option should be selected first for each Excel Worksheet.



Soil concentration (pCi/g)	200.0				
Risk	Exposure Pathway				Total
by Radionuclide	Inhalation	Soil	Food	External	Risk
Am-241	2.23E-07	7.78E-07	1.76E-06	5.67E-06	8.4E-06
Pu-239	2.64E-07	9.93E-07	1.36E-07	4.11E-08	1.4E-06
Ur-234	9.03E-08	5.66E-07	7.51E-07	5.18E-08	1.5E-06
Ur-235	8.00E-08	5.84E-07	7.68E-07	1.12E-04	1.1E-04
Ur-238	7.41E-08	7.53E-07	9.52E-07	2.34E-05	2.5E-05

> For calculation of Risks, input soil concentration (cell C5)
> For calculation of PPRGs, input Target Risk (cell J4)

Because pathway-specific calculations are given, the spreadsheets can also be used to calculate the percent contribution to the total risk (or RSAL). The total contribution is a function of both the exposure and toxicity variables for each radionuclide. Figure C-2 displays an example of the results for the Rural Resident scenario. It should be noted that since the percent contribution is independent of the concentration in soil, the results will be the same regardless of whether the spreadsheet is used to calculate risk or RSAL. The equations are set up to track the percent contributions for the forward-facing calculations of risk.

Risk by RAD	% Total by Exposure Pathway				Total
	Inhalation	Soil	Food	External	%
Am-241	2.6%	9.2%	20.9%	67.3%	100%
Pu-239	18.4%	69.2%	9.5%	2.9%	100%
Ur-234	6.2%	38.8%	51.5%	3.5%	100%
Ur-235	0.1%	0.5%	0.7%	98.7%	100%
Ur-238	0.3%	3.0%	3.8%	92.9%	100%

Figure C-2. Results showing the percent contribution of exposure pathway by radionuclide. The total sums to 100 percent for each radionuclide. The example is from one iteration of a Monte Carlo simulation using the Excel worksheet for the Rural Resident exposure scenario.

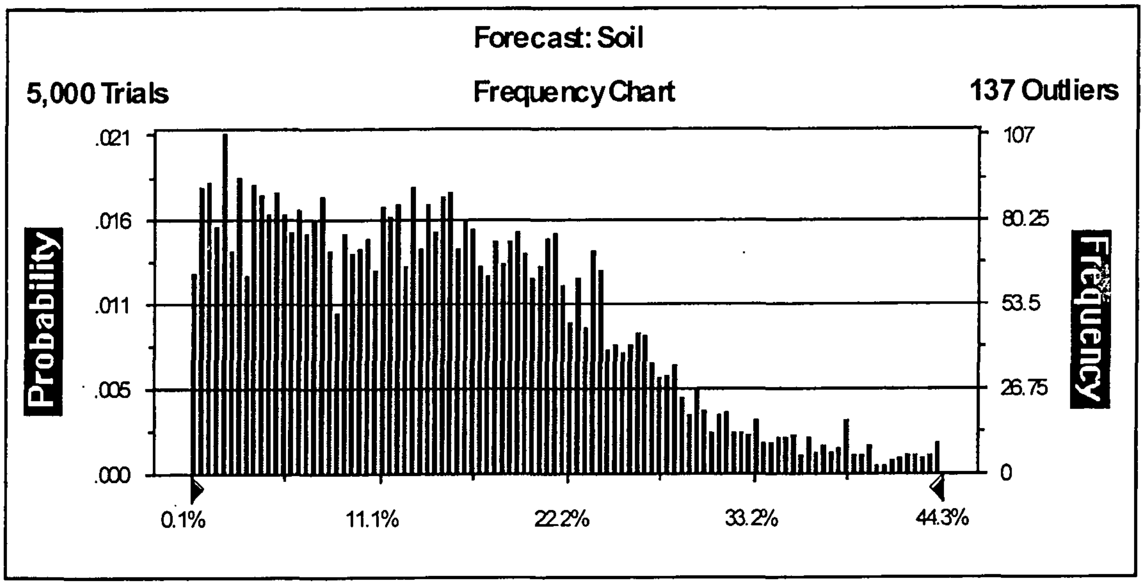


Figure C-3. Results of a Monte Carlo simulation with 5,000 iterations showing the probability distribution for the percent contribution of the soil ingestion pathway to total Am-241 Risk under the Rural Resident scenario. The average contribution of the soil pathway is approximately 16 percent, while the 95th percentile is approximately 37 .

For the point estimate calculation, one set of final results will be displayed in the output range (e.g., cells O6 : R10). However, for the probabilistic simulations, one set of results represents one iteration (or trial) of the Monte Carlo simulation. If a Monte Carlo simulation is run with 5,000 trials, the calculations will be repeated 5,000 times. Therefore, when the worksheet is first opened, the numbers displayed for the "probabilistic results" should be interpreted with caution. Each cell in this range can be tracked as a "forecast cell", as discussed below, so that summary statistics can be obtained after the simulation has ended. Figure C-3 gives the probability distribution of percent contribution for the soil ingestion pathway for Am-241 under the Rural Resident scenario. In this example, one would conclude that the average percent contribution of soil to the total risk of Am-241 is 16 percent, however, the 95th percentile is 37 percent. These means that there is a 5 percent probability that soil contributes more than one third to the total risk of Am-241 for the Rural Resident population.

B. Crystal Ball Settings and Instructions

Instructions for obtaining both point estimate results and probabilistic results are given in each Excel worksheet. An example for the Rural Resident scenario is given in Table C-2. The difference between the point estimate and probabilistic approaches is that under the point estimate approach, all of the input variables are described by a single fixed values, whereas the probabilistic results use a probability distribution for one or more input variables. The same set of equations are used in both approaches.

In order to run the Monte Carlo analysis with these worksheets, the following software was used: Crystal Ball 2000 Professional Edition version 5.1 (Decisioneering, http://www.decisioneering.com), Microsoft Excel 2000, and a Windows 98 operating system. While this appendix provides highlights of the steps required to run a Monte Carlo simulation, it is not intended to be a comprehensive tutorial or substitute for professional training classes in Monte Carlo analysis or probabilistic risk assessment (PRA).

Steps 7-14 of the Instructions given in Table C-2 provide a step-by-step guide to running a Monte Carlo simulation. It is highly recommended that one open a worksheet after having opened Crystal Ball. By opening Crystal Ball, Excel will automatically open as well. Choose to enable the macros when prompted. After the spreadsheet is successfully opened, the important components of running an analysis can be divided into 4 major areas: 1) Specifying probability distributions for one or more input variables; 2) Inputting the Settings to run a Monte Carlo analysis; 3) Specifying the cells that contain the output of interest; and 4) Running the simulation. Table C-2 provides instructions for using the Crystal Ball commands given in the pull-down menus of the toolbar. Some of the same commands can be executed by using the short-cut icons in the toolbar that is added to the desktop after Crystal Ball is opened (see Figure C-4).

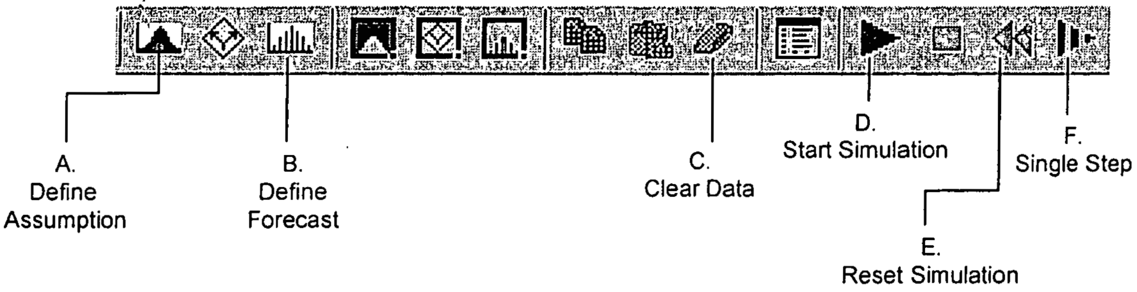


Figure C-4. Crystal Ball's toolbar of short-cut icons that are added to the Microsoft Excel toolbar. The following describes the function and purpose of each icon.

A. Define Assumption – used to define the type of probability distribution and the parameter values for the distribution. First, click on the "place holder" cell in **Column F,** then click this icon to view the distribution options. If a distribution is already assigned, you will see a graph of the distribution, and references to cells on the spreadsheet that define the parameters (i.e., Columns G: K. If a distribution is not yet assigned, you will see a Gallery of options. In each worksheet, pre-defined cells are highlighted with green shading.

B. Define Forecast – used to indicate which cell(s) to track during a Monte Carlo simulation in order to present a distribution of results. Options include: risk estimates, PPRG estimates, and percent contributions of exposure pathways by radionuclide.

C. Clear Data – will remove a definition of either an assumption (A) or a forecast (B). Simply select the cell, and click on the icon. Crystal Ball will prompt the user to delete the definitions.

D. Start Simulation – used to run a simulation after the run preferences have been defined.

E. Reset Simulation – used to reset the Crystal Ball simulation to rerun a new simulation. This option should ALWAYS be selected for consecutive simulations.

F. Single Step – used to run one iteration. This is a useful feature to verify that random values are being selected for the desired cells in a spreadsheet. It has a similar utility to the F9 key (Recalculate) in Excel.

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Table C-2. Example of "Instructions Sheet" provided for the Rural Resident exposure scenario.

Instructions for Using Excel Spreadsheets to Caculate Risk or PPRG with U.S. EPA Standard Risk Equations

Step	Description	Action
1	To begin, open the spreadsheet, "Residential"	Click on the name at the bottom of this spreadsheet
2	Select type of calculation - point estimate or probabilistic	Click on 1 of 2 options in the dialogue box at the top of columns F & G
If calculating point estimates, go to Steps 3-6. If calculating probabilistic estimates, skip to Steps 7-14.		
3	Point Estimate Inputs - exposure and toxicity	Change values for exposure variables in Column E Change values for dose-response in "Toxicity" tab
4	Risk calculation	Enter soil concentration in Cell C3. This value will apply equally to all radionuclides
5	PRG calculation	Enter a target risk in Cell J4. This value will apply equally to all radionuclides
6	Results - Risk, PRG, % by Pathway	Risk estimates are given in cells G6: G10 PRG estimates are given in cells **J6: J10** % by exposure pathway are given in cells **O6: R10**; these results apply equally to the risk or PRG calculations
7	Monte Carlo simulations	Crystal Ball (CB) is needed to run Monte Carlo simulations. If CB is not open, exit Excel, open CB, and open this spreadsheet.
8	Enter Probability Distribution Functions (PDFs) by Defining Assumptions	CB has a separate menu for inputting distributions. CB requires a unique cell for each assignment of a distribution. **Column F**, called "Values", has been reserved for this purpose. Cells that are defined as PDFs are shaded "green", whereas cells that are defined as point estimates have no shading. The definition of the PDF is given in the adjacent cells in **Columns G:K**. To change parameter values, simply change the values in **Columns H: K**. To change both the distribution type and parameter values, click on the cell in **Column F**, and choose "Cell / Define Assumptions" from the menu bar, then select Gallery
9	Choose Results to Track	Results that may be of interest: risks, PRGs, % contribution by pathway. Be sure to select the "Probabilistic results" from the toggle in columns F&G (See Step 2). > Risk estimates are given in cells **G6: G10** > PRG estimates are given in cells **K6: K10** > % by exposure pathway are given in cells **O6: R10**
10	Define Forecasts	Before running a Monte Carlo simulation, you need to identify which output cells to track. Click on the cell you want to track from among the options in Step 9. Choose "Cell / Define Forecasts" from the menu bar. Enter a unique name for the forecast cell (e.g., Am-241 Risk). Repeat for each Forecast cell.
11	Monte Carlo simulation settings: number of trials, sampling	Choose these settings prior to running the first Monte Carlo simulation. Options are located in "Run / Run preferences". Click on Trials to set the number of trials (or iterations); Click on Sampling to set the sampling to Latin Hypercube. Click on Speed and select options as desired to increase the sampling speed.
12	Run a Monte Carlo Simulation	After the settings have been selected (see Step 11), run a simulation by clicking on the solid green arrow that points to the right on the menu bar, or choose "Run / Run". To Rerun a simulation, it is import to RESET Crystal Ball. Do this by clicking on the broken green arrow that points to the left on the menu bar.
13	View Results	CB provides the following results automatically after a simulation is complete: a graph showing the distribution of results; summary statistics in increments of 10th percentiles. A report can be generated by choosing "Run / Create Report". Additional percentiles can be obtained. If the statistic of interest is not generated by this report, the data must be exported to Excel and calculated manually within Excel. Export data by choosing, "Run / Extract Data"
14	Obtain Exact Results	Every time a Monte Carlo simulation is run, values are selected at random from the probability distributions defined as assumption cells. Repeating simulations with the same number of iterations will give similar, but not exactly the same results. To obtain exactly reproducible results, it is necessary to fix the random number seed and note all of the settings. This option is available in " Run / Run Preferences" then click on Sampling, and click on the box for "Use the same Sequence of Random Numbers" and pick any value for the seed. ***NOTE - this option will work for only the 1st simulation after opening CB. Therefore, first close out of CB, then reopen CB and the spreadsheet.

Viewing Results

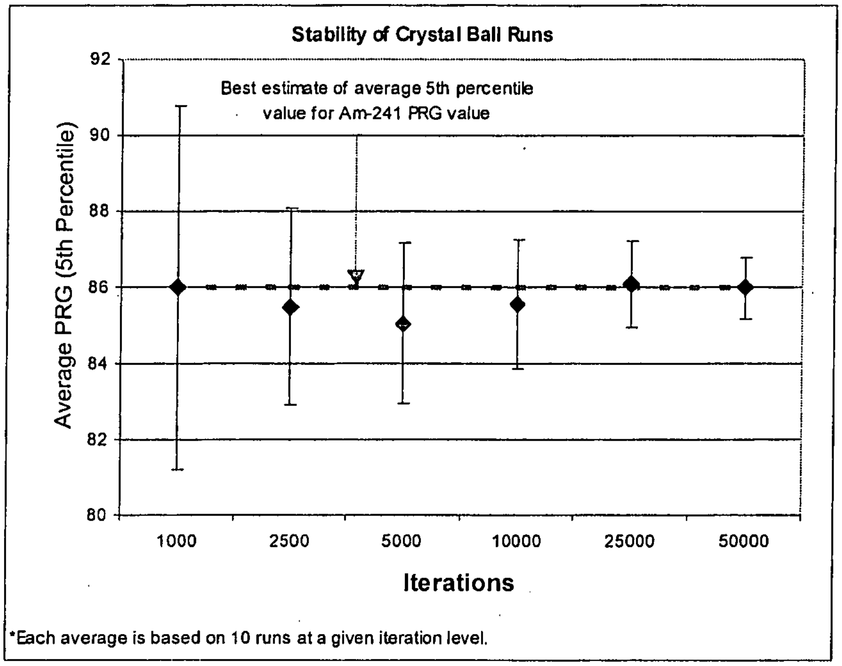
When a simulation completes, Crystal Ball will display results of the forecasts automatically, unless this feature is disabled. If results are not displayed, choose "Run / Forecast Windows / Open all Forecasts". Crystal Ball provides a variety of automated output, including graphs of the forecast cells (both the PDF and CDF views), a slider button on the graphs to obtain different percentile estimates, and summary statistics tables with the mean, standard deviation and selected percentiles. If Crystal Ball's output does not provide the desired summary, the raw data from each iteration can be exported to a new Excel sheet ("Run / Export Data"), where a separate data analysis can be performed.

Stability of the Output Distributions

The goal of a Monte Carlo simulation is to provide a reasonable approximation of the output distribution, given a set of input distributions and an algebraic equation for risk or PPRG. Different numbers of iterations (referred to by Crystal Ball as trials) may be needed, depending on the characteristics of the input distributions, the form of the equation, and the statistics of interest in the output distribution. In general, statistics nearer to the tails of the output distribution (e.g., 5th or 95th percentiles) are less stable than statistics that describe the central tendency (e.g., arithmetic mean, 50th percentile). For the risk equations and distributions used in this analysis, sufficient stability can be obtained with 10,000 iterations. Examples are given for the 1st and 5th percentiles of the distribution of PPRGs for Am-241 in Figure C-5. One standard deviation differs from the mean by only 2 percent for the 5th percentile and 5 percent for the 1st percentile based on 10 repeated simulations.

Reproducing Results Exactly

Sometimes it may be desirable to run a simulation that can be reproduced exactly. This is a useful feature for regulatory review or QA/QC of probabilistic models, for example. The following settings would need to be reported in order to reproduce simulation results exactly: worksheet, software used, forecast cell, number of trials of the Monte Carlo simulation, random number seed, and sampling type (i.e., Monte Carlo or Latin Hypercube). This feature was not employed for the simulation results reported in this report. However, each of the worksheets do allow for this feature to be activated by selecting the "Run / Run Preferences" option in Crystal Ball.



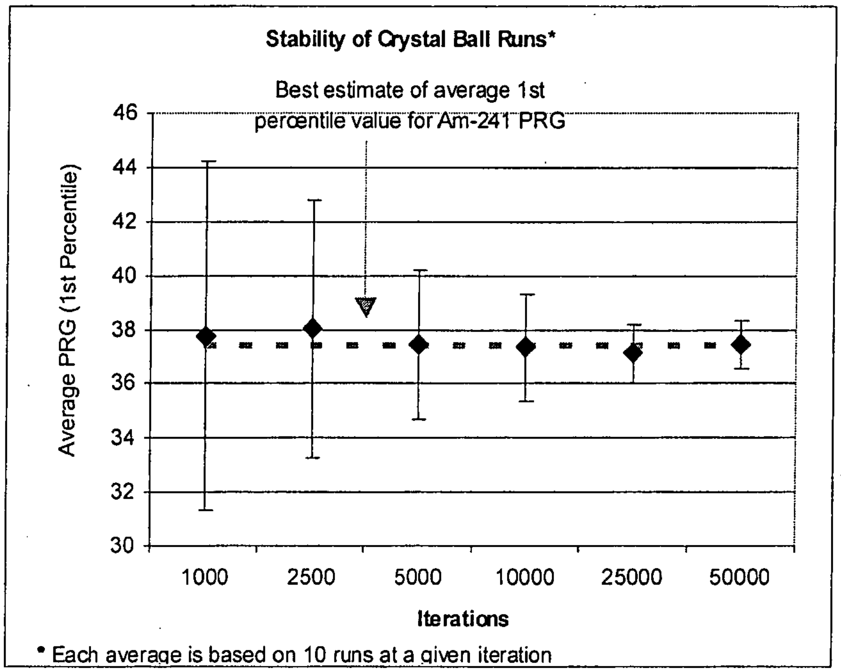


Figure C-5. Results of stability evaluations for Monte Carlo simulations using the PPRG for Am-241 and a Target Risk of 1E-06 as an example. The top graph illustrates the mean and standard deviation 5th percentile PPRG for n = 10 simulations for different numbers of iterations, with the "best estimate" equal to the mean for 50,000 iterations. The bottom graph illustrates the same information but for the 1st percentile PPRG.

APPENDIX D - COMPLETE RESRAD INPUT PARAMETERS FOR DOSE CALCULATIONS

Computer modeling of environmental radiation exposure involves considerable simplification, mathematically, of a complex system. This simplification can be justified, if it can be demonstrated that the computer model gives similar results to other accepted models, or that it can be verified to accurately or at least, conservatively predict results which can be measured in real environmental systems. The RESRAD computer model has the advantages of being easy to use, well documented, and successfully tested against other models and against several real systems (Yu et al, 2001, Chapter 5). The power of the RESRAD 6.0 model resides not only in its extensive libraries of radionuclide data, dose conversion factors, and default values for parameters, but in its user friendly interface and ability to handle parameters input as distributions, and to perform Monte Carlo dose and risk analyses and uncertainty analyses. For all its impressive features, RESRAD 6.0 is mathematically a very simple model, especially for the pathway calculations that are relevant at Rocky Flats. The degree of simplicity inherent in RESRAD is the result of the simplifying assumptions about the environmental system modeled, and these assumptions, in turn, affect the degree of detail in scenario features and parameter values which can be addressed by RESRAD.

The primary simplifications inherent in RESRAD include the following:

- The contaminated zone is circular in shape with the receptor in the center, but can be modified by a user specified shape factor.

- The residual contamination is of uniform concentration (highest value less than 3 times the mean value, lowest greater than 1/3 the mean value). This is an appropriate and even conservative assumption for a site that has been cleaned up to the RSAL value.

- For areas of contamination greater than 1000 m2 (20,000 m2 for meat and milk) all pathways except the inhalation pathway are independent of area (saturated). Because of this, and the assumption of uniform contamination, specific location of a receptor on a large cleaned up site (like ROCKY FLATS in the future) would be unimportant, since the exposure rate would be fairly uniform over the whole site.

- For the inhalation pathway, a simple "box model", modified by an area and wind speed dependent dilution factor is assumed. While this would be considered an inappropriate tool for short term transport modeling, it has been shown to be adequate for approximating dose due to average exposure conditions over 1 year periods. Under such circumstances the fluctuations in wind direction tend to average out, and the receptor is exposed to contaminated dust at close to the value of average mass loading which is the input parameter required by the model.

- For the inhalation pathway, the value of annual average mass loading is assumed to be present as respirable particles only (1 micron AMAD). This is generally a conservative assumption, since the use of site specific data (PM10 or TSP) as a surrogate for 1 micron particulates would overestimate.

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Table D-1 summarizes the full list of pathways and input parameter values that were used for each scenario modeled using RESRAD 6.0 with a 25 millirem per year dose limit. Scenarios typically differ from one another in terms of only a few parameters (see, for example, breathing rates, indoor/outdoor time fractions, soil and plant ingestion rates, etc.). This is because most of the input parameters are physical features of the site being evaluated and are usually the same for all scenarios.

The RESRAD default parameters and values used in the 1996 computation of RSALS for the residential scenario are also displayed in Table D-1. Note that the 1996 computation used an earlier version of RESRAD which did not use the "area correction factor" to adjust the inhalation pathway dose for dilution, and 85 and 15 mrem/yr dose limits, so the results are not directly comparable to the results of this task.

The pathway and parameter data are presented in the order in which RESRAD prompts the user for inputs. Most of the information in Table D-1 is straightforward, however, several conventions warrant explanation. In the pathway section, the terms "active" and "suppressed" refer to whether the pathway calculation is turned on or off, respectively, a feature of RESRAD that makes it adaptable to a wide variety of situations.

The term "not used" appears throughout the table. This term is applied in some situations when an option is not applicable (for example Time for Calculations). In other situations it is applied automatically when the given parameter is required but the pathway is turned off.. In some cases an input parameter value and "not used" appear together. In these cases, the value of the input parameter would be as specified if the pathway was turned on.

For parameters that are input as fixed values, a single number is given. For parameters that are input to RESRAD 6.0 as distributions, the convention is to specify the base value (type of distribution, parameters that describe the distribution) in bold type. For example, Inhalation Rate for rural resident (adult) is presented as 8400(log norm-N: 8.657, 0.237). This means the first number, 8400, signifies the single value for this parameter selected by the working group. The data in parentheses is information about the distribution that the user is prompted to provide as input parameters for RESRAD.

RESRAD 6.0 permits the use of "continuous linear" parameter values, limited to 8 total data pairs for any distributed parameter, to enable the use of empirical data. What is the significance of this compared to other parameters? For the two distributions for mass loading (for inhalation and for foliar deposition) designated as "PDF # 1" and "PDF # 2", the values of the 8 data points used to define each distribution are presented at the very end of Table D-1.

Table D-1 Input Parameters for All Scenarios

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 Input VALUE	Rural Resident (ADULT)	Rural Resident (CHILD)	Wildlife Refuge Worker	Office Worker	Open Space User
Pathways								
External Gamma		active	active	active	active	active	active	active
Inhalation		active	active	active	active	active	active	active
Plant Ingestion		active	active	active	active	Suppressed	Suppressed	Suppressed
Meat Ingestion		active	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed
Milk Ingestion		active	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed
Aquatic Foods		active	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed
Drinking Water		active	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed
Soil Ingestion		active	active	active	active	active	active	active
Radon		active	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed	Suppressed
Initial Principal Radionuclide Activity in Contaminated Zone	pCi/g		Am-241	100	100	100	100	100
	pCi/g		Pu-238					
	pCi/g		Pu-239	100	100	100	100	100
	pCi/g		Pu-240					
	pCi/g		Pu-241					
	pCi/g		Pu-242					
Basic Radiation Dose Limit	mrem/y	25	15	25	25	25	25	25
Time for Calculations	y	1	0.2	1	1	1	1	1
Time for Calculations	y	3	1	3	3	3	3	3
Time for Calculations	y	10	5	10	10	10	10	10
Time for Calculations	y	30	not used	30	30	30	30	30
Time for Calculations	y	100	not used	100	100	100	100	100
Time for Calculations	y	300	not used	300	300	300	300	300
Time for Calculations	y	1000	not used	1000	1000	1000	1000	1000
Time for Calculations	y	not used	not used	not used	not used	not used	not used	not used
Time for Calculations	y	not used	not used	not used	not used	not used	not used	not used

Occupancy, Inhalation, and External Gamma								
Inhalation Rate	m^3/y	8400	7000	8400 (log norm-N: 8.657, 0.237)	5256 (log norm-N: 8.084, 0.305)	14000(Beta: 9636, 17560, 1.79, 3.06)	9636	14852
Mass Loading for Inhalation	g/m^3	0.0001	0.000026	0.000058 (PDF 1)	0.000058 (PDF 1)	0.000058 (PDF 1)	0.000058	0.000058
Exposure Duration	y	30	30	30 not used	30 not used	30 not used	30 not used	30 not used
Indoor Dust Filtration Factor		0.4	1	0.7	0.7	0.7	0.4	0.7
External Gamma Shielding Factor		0.7	0.8	0.4	0.4	0.4	0.4	0.4
Indoor Time Fraction		0.5	1	0.82 (triangular .408; .545; .815)	0.82 (triangular .408; .545; .815)	0.114(B-Norm: .103; .005; .091; .114)	0.23	0
Outdoor Time Fraction		0.25	0	0.14 (triangular .072; .096; .144)	0.14 (triangular .072; .096; .144)	0.114(B-Norm: .103; .005; .091; .114)	0	0.03
Shape Factor for external gamma		1	1	1	1	1	1	1
Area of Contaminated Zone	m^2	10000	40000	1,400,000	1,400,000	1,400,000	1,400,000	1,400,000
Thickness of Contaminated Zone	m	2	0.15	0.15	0.15	0.15	0.15	0.15
Length Parallel to Aquifer Flow	m	100	200	200	200	200	200	200
RESRAD 6.0 INPUT PARAMETERS	**UNITS**	**RESRAD 6.0 Default**	**1996 VALUE**	**Rural Resident (ADULT)**	**Rural Resident (CHILD)**	**Wildlife Refuge Worker**	**Office Worker**	**Open Space User**
Cover and Contaminated Zone Hydrological Data								
Cover Depth	m	0	not used	No Cover	No Cover	No Cover	No Cover	No Cover
Density of Cover Material	g/cm^3	1.5	not used	No Cover	No Cover	No Cover	No Cover	No Cover
Cover Erosion Rate	m/y	0.001	not used	No Cover	No Cover	No Cover	No Cover	No Cover
Density of Contaminated Zone	g/cm^3	1.5	1.8	1.7	1.7	1.7	1.7	1.7
Contaminated Zone Erosion Rate	m/y	0.001	0.0000749	0.0000749	0.0000749	0.0000749	0.0000749	0.0000749

Contaminated Zone Total Porosity		0.4	0.3	0.3	0.3	0.3	0.3	0.3
Contaminated Zone Field Capacity		0.2	0.1	0.1	0.1	0.1	0.1	0.1
Contaminated Zone Hydraulic Conductivity	m/y	10	44.5	44.5	44.5	44.5	44.5	44.5
Contaminated Zone b Parameter		5.3	10.4	10.4	10.4	10.4	10.4	10.4
Humidity in Air	g/m^3	8	not used	not used	not used	not used	not used	not used
Evapotranspiration Coefficient		0.5	0.253	0.253	0.253	0.253	0.253	0.253
Average Annual Wind Speed	m/s	2	2	4.2	4.2	4.2	4.2	4.2
Precipitation	m/y	1	0.381	0.381	0.381	0.381	0.381	0.381
Irrigation	m/y	0.2	1	1	1	0	0	0
Irrigation Mode		overhead	overhead	overhead	overhead	overhead	overhead	overhead
Runoff Coefficient		0.2	0.004	0.004	0.004	0.004	0.004	0.004
Watershed Area	m^2	1E+06	8280000	8280000	8280000	8280000	8280000	8280000
Accuracy for Water/Soil Computations		0.001	0.001	0.001	0.001	0.001	0.001	0.001
Uncontaminated Unsaturated Zone Parameters								
Number of Unsaturated Zone Strata		1	1	1	1	1	1	1
Thickness	m	4	3	3	3	3	3	3
Density	g/cm^3	1.5	1.8	1.7	1.7	1.7	1.7	1.7
Total Porosity		0.4	0.3	0.3	0.3	0.3	0.3	0.3
Effective Porosity		0.2	0.1	0.1	0.1	0.1	0.1	0.1
Field Capacity		0.2	0.1	0.1	0.1	0.1	0.1	0.1
Hydraulic Conductivity	m/y	10	44.5	44.5	44.5	44.5	44.5	44.5
b Parameter		5.3	10.4	10.4	10.4	10.4	10.4	10.4
Radionuclide Transport Factors								
Distribution Coefficient Contaminated Zone	cm^3/g	-		Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800	Pu = 2300, Am = 1800
Distribution Coefficient Unsaturated Zone	cm^3/g	-		Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800	Pu = 2300, Am = 1800
Distribution Coefficient Saturated Zone	cm^3/g	-		Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800, U = 2.3	Pu = 2300, Am = 1800	Pu = 2300, Am = 1800
Time since placement of materials	year	0		0	0	0	0	0
Solubility Limit	mol/l	0		0	0	0	0	0
Leach Rate	$year^{-1}$	0		0	0	0	0	0

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 VALUE	Rural Resident (ADULT)	Rural Resident (CHILD)	Wildlife Refuge Worker	Office Worker	Open Space User
Saturated Zone Hydrological Data								
Density of Saturated Zone	g/cm^3	1.5	1.8	1.7 - not used	1.7 - not used	1.7 - not used	1.7 - not used	1.7 - not used
Saturated Zone Total Porosity		0.4	0.3	0.3 - not used	0.3 - not used	0.3 - not used	0.3 - not used	0.3 - not used
Saturated Zone Effective Porosity		0.2	0.1	0.1 - not used	0.1 - not used	0.1 - not used	0.1 - not used	0.1 - not used
Saturated Zone Field Capacity		0.2	0.1	0.1 - not used	0.1 - not used	0.1 - not used	0.1 - not used	0.1 - not used
Saturated Zone Hydraulic Conductivity	m/y	100	44.5	44.5 - not used	44.5 - not used	44.5 - not used	44.5 - not used	44.5 - not used
Saturated Zone Hydraulic Gradient		0.02	0.15	0.15 - not used	0.15 - not used	0.15 - not used	0.15 - not used	0.15 - not used
Saturated Zone b Parameter		5.3	not used	10.4 - not used	10.4 - not used	10.4 - not used	10.4 - not used	10.4 - not used
Water Table Drop Rate		0.001	0	0 - not used	0 - not used	0 - not used	0 - not used	0 - not used
Well Pump Intake Depth (below water table)	m	10	10	10 - not used	10 - not used	10 - not used	10 - not used	10 - not used
Model: nondispersion (ND) or Mass-Balance (MB)		ND	ND	ND - not used	ND - not used	ND - not used	ND - not used	ND - not used
Well Pumping Rate	m^3/y	250	250	250 - not used	250 - not used	250 - not used	250 - not used	250 - not used
Ingestion Pathway, Dietary Data								
Fruit, Vegetable and Grain Consumption	kg/y	160	40.1	85 (Log norm-N: 3.566, 1.446)	42.5 (Log norm-N: 2.024, 1.042)	not used	not used	not used
Leafy Vegetable Consumption	kg/y	14	2.6	6.4 (Log norm-N: 0.418, 1.783)	3.2 (Log norm-N: -1.122, 1.775)	not used	not used	not used
Milk Consumption	l/y	92	not used	not used	not used	not used	not used	not used
Meat and Poultry Consumption	kg/y	63	not used	not used	not used	not used	not used	not used
Fish Consumption	kg/y	5.4	not used	not used	not used	not used	not used	not used
Other Seafood Consumption	kg/y	0.9	not used	not used	not used	not used	not used	not used

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 VALUE	Rural Resident (ADULT)	Rural Resident (CHILD)	Wildlife Refuge Worker	Office Worker	Open Space User
Soil Ingestion	g/y	36.5	70	36.5	70 (B-Log norm-N: 1.912, 1.371, 1, 365)	109.5	36.5	36.5
Drinking Water Intake	l/y	510	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Drinking Water		1	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Household Water		1	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Livestock Water		1	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Irrigation Water		1	0	0	0	0	0	0
Contaminated Fraction, Aquatic Food		0.5	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Plant Food		-1	1	1	1	1	1	1
Contaminated Fraction, Meat		-1	not used	not used	not used	not used	not used	not used
Contaminated Fraction, Milk		-1	not used	not used	not used	not used	not used	not used
Ingestion Pathway, Nondietary Data								
Livestock Fodder Intake For Meat	kg/day	68	not used	not used	not used	not used	not used	not used
Livestock Fodder Intake for Milk	kg/day	55	not used	not used	not used	not used	not used	not used
Livestock Water Intake For Meat	l/d	50	not used	not used	not used	not used	not used	not used
Livestock Water Intake For Milk	l/d	160	not used	not used	not used	not used	not used	not used
Livestock Intake For Soil	kg/day	0.5	not used	not used	not used	not used	not used	not used
Mass Loading for Foliar Deposition	g/m^3	0.0001	0.0001	0.000145	0.000145	not used	not used	not used
Depth of Soil Mixing Layer	m	0.15	0.15	0.15	0.15	0.15	0.15	0.15
Depth of Roots	m	0.9	0.9	0.15	0.15	not used	not used	not used
Groundwater Fractional Usage, Drinking Water		1	not used	not used	not used	not used	not used	not used
Groundwater Fractional Usage, Household Water		1	not used	not used	not used	not used	not used	not used
Groundwater Fractional Usage, Livestock Water		1	not used	not used	not used	not used	not used	not used
Groundwater Fractional Usage, Irrigation Water		1	not used	not used	not used	not used	not used	not used

Plant Factors								
Wet Weight Crop Yield, Non-Leafy	kg/m^2	0.7	0.7	0.7	0.7	not used	not used	not used
Length of Growing Season, Non-Leafy	years	0.17	0.17	0.17	0.17	not used	not used	not used
Translocation Factor, Non-Leafy		0.1	0.1	0.1	0.1	not used	not used	not used
Weathering Removal Constant	1/year	20	20	20	20	not used	not used	not used
Wet Foliar Interception Fraction, Non-Leafy		0.25	0.25	0.25	0.25	not used	not used	not used
Dry Foliar Interception Fraction, Non-Leafy		0.25	0.25	0.25	0.25	not used	not used	not used
Wet Weight Crop Yield, Leafy	kg/m^2	1.5	1.5	1.5	1.5	not used	not used	not used
Length of Growing Season, Leafy	years	0.25	0.25	0.25	0.25	not used	not used	not used
Translocation Factor, Leafy		1	1	1	1	not used	not used	not used
Wet Foliar Interception Fraction, Leafy		0.25	0.25	0.25	0.25	not used	not used	not used
Dry Foliar Interception Fraction, Leafy		0.25	0.25	0.25	0.25	not used	not used	not used
Wet Weight Crop Yield, Fodder	kg/m^2	1.1	1.1	1.1	1.1	not used	not used	not used
Length of Growing Season, Fodder	years	0.08	0.08	0.08	0.08	not used	not used	not used
Translocation Factor, Fodder		1	1	1	1	not used	not used	not used
Weathering Removal Constant, Fodder	1/year	20	20	20	20	not used	not used	not used
Wet Foliar Interception Fraction, Fodder		0.25	0.25	0.25	0.25	not used	not used	not used
Dry Foliar Interception Fraction, Fodder		0.25	0.25	0.25	0.25	not used	not used	not used

Storage Times Before Use Data								
Fruits, Non-Leafy Vegetables and Grain	days	14	14	14	14	not used	not used	not used
Leafy Vegetables	days	1	1	1	1	not used	not used	not used
Milk	days	1	1	not used	not used	not used	not used	not used
Meat	days	20	20	not used	not used	not used	not used	not used
Fish	days	7	7	not used	not used	not used	not used	not used
Crustacea and Mollusks	days	7	7	not used	not used	not used	not used	not used
Well Water	days	1	1	not used	not used	not used	not used	not used
Surface Water	days	1	1	not used	not used	not used	not used	not used
Livestock Fodder	days	45	45	not used	not used	not used	not used	not used

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PDF # 1 *** Continuous Linear ***	Value	cdf
Mass Loading for Inhalation	10	0
units are micrograms/m^3	20.2	0.338
	23.1	0.788
	50.7	0.919
	58	0.944
	95.7	0.969
	109	0.994
	200	1

PDF # 2 ** Continuous Linear **	Value	cdf
Mass Loading for Foliar Deposition units are micrograms/m^3	25	0
	50.5	0.338
	57.7	0.788
	127	0.919
	145	0.944
	239	0.969
	274	0.994
	500	1

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Appendix E - RESRAD Run Results Printout

A CD-ROM with this information is available from the Department of Energy upon request

Appendix F - PM-10 Air Monitoring Data from Rocky Flats and the State of Colorado

Rocky Flats Specific Data

all monitor values in micrograms per cubic meter ($\mu g/m^3$)	Year	No. of 24-hr Values	1st Max of 24-hr Values	2nd Max of 24-hr Values	3rd Max of 24-hr Values	4th Max of 24-hr Values	Annual Mean
Location							
X-1							
	1995	57	31	25	22	21	9.7
	1996	60	31	30	28	23	11.7
	1997	58	25	23	22	18	9.4
	1998	59	33	26	20	20	10.7
	1999	55	25	25	19	19	10.1
	2000	35	30	27	24	21	11.3
Max		60	33	30	28	23	11.7
X-2							
	1995	59	34	26	24	24	11.5
	1996	60	32	29	28	28	13
	1997	58	25	23	22	19	10.7
	1998	61	33	27	21	21	12
	1999	57	29	24	23	23	11.3
	2000	60	29	26	25	25	12.8
Max		61	34	29	28	28	13
X-3							
	1995	54	87	57	46	39	16.6
	1996	59	32	28	26	26	13.1
	1997	61	25	24	21	20	10.6
	1998	59	33	27	25	21	12.2
	1999	53	47	28	26	21	11.6
	2000	61	28	24	24	22	12.5
Max		61	87	57	46	39	16.6
X-4							
	1995	55	34	26	25	21	11
	1996	56	36	29	28	25	13.7
	1997	59	23	20	19	18	10.1
	1998	60	33	25	21	21	11.2
	1999	52	26	24	21	18	9.7
	2000	60	27	24	23	22	11.7
Max		60	36	29	28	25	13.7
X-5							
	1995	57	37	31	28	25	12.3
	1996	60	41	39	33	32	14.7
	1997	57	26	23	21	21	11.3
	1998	56	33	26	23	23	12.4
	1999	53	31	29	26	23	11.6
	2000	55	27	26	25	24	13
Max		60	41	39	33	32	14.7

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Colorado PM-10 Data from EPA's AIRS Database

(Monday, 28-Jun-1999 at 6:4:20 PM (USA Eastern time zone)

No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
359	179	142	135	114	1	1	38.3	1993		Adams	CO
347	122	107	99	87	0	0	35.9	1994		Adams	CO
344	99	97	88	88	0	0	33.1	1995		Adams	CO
350	98	96	90	82	0	0	33.6	1996		Adams	CO
345	98	98	96	94	0	0	34.8	1997		Adams	CO
344	118	99	93	86	0	0	36.1	1998		Adams	CO
61	73	72	70	68	0	0	25.6	1993	Northglenn	Adams	CO
59	86	40	39	38	0	0	23.5	1994	Northglenn	Adams	CO
48	41	37	36	34	0	0	21	1995	Northglenn	Adams	CO
148	82	73	68	67	0	0	26.5	1993	Brighton	Adams	CO
160	68	61	55	50	0	0	22.5	1994	Brighton	Adams	CO
174	101	84	46	46	0	0	20.5	1995	Brighton	Adams	CO
147	57	54	52	48	0	0	23.3	1996	Brighton	Adams	CO
112	86	71	58	54	0	0	23.3	1997	Brighton	Adams	CO
114	64	55	51	47	0	0	21.2	1998	Brighton	Adams	CO
114	83	77	76	75	0	0	26.9	1993		Adams	CO
114	90	53	52	48	0	0	23.6	1994		Adams	CO
113	73	46	42	40	0	0	21	1995		Adams	CO
111	59	57	48	47	0	0	21	1996		Adams	CO
128	60	46	44	44	0	0	21.8	1997		Adams	CO
58	40	39	37	37	0	0	21.9	1998		Adams	CO
351	80	61	60	52	0	0	17.7	1993		Adams	CO
351	54	51	50	47	0	0	17.1	1994		Adams	CO
301	55	44	36	35	0	0	16.5	1995		Adams	CO
340	59	58	46	44	0	0	19.4	1996		Adams	CO
265	59	53	45	45	0	0	17.2	1997		Adams	CO
326	62	56	50	45	0	0	19.3	1998		Adams	CO
342	99	69	68	64	0	0	24.7	1993	Alamosa	Alamosa	CO
345	88	83	71	68	0	0	22.9	1994	Alamosa	Alamosa	CO
350	125	86	79	72	0	0	22.4	1995	Alamosa	Alamosa	CO
309	127	92	91	69	0	0	21.3	1996	Alamosa	Alamosa	CO
332	144	113	110	93	0	0	21.6	1997	Alamosa	Alamosa	CO
333	101	88	81	72	0	0	22.9	1998	Alamosa	Alamosa	CO
61	98	98	75	65	0	0	29.4	1993	Englewood	Arapahoe	CO
59	61	60	54	49	0	0	24.3	1994	Englewood	Arapahoe	CO
14	43	33	31	31	0	0	24.9	1995	Englewood	Arapahoe	CO
339	126	125	124	113	0	0	43.5	1993		Archuleta	CO
346	262	258	110	109	2	2	41.1	1994		Archuleta	CO
335	98	97	83	80	0	0	31.7	1995		Archuleta	CO
351	85	85	78	77	0	0	32	1996		Archuleta	CO
339	120	96	89	85	0	0	29.2	1997		Archuleta	CO
335	66	66	64	61	0	0	27.2	1998		Archuleta	CO
55	75	65	61	52	0	0	23.5	1993	Boulder	Boulder	CO
55	37	35	32	32	0	0	16.9	1994	Boulder	Boulder	CO
54	35	29	23	22	0	0	13.1	1995	Boulder	Boulder	CO
59	41	31	28	26	0	0	15.8	1996	Boulder	Boulder	CO
43	28	27	24	24	0	0	15.2	1997	Boulder	Boulder	CO
334	98	81	72	66	0	0	25	1993	Longmont	Boulder	CO
330	72	58	51	49	0	0	21	1994	Longmont	Boulder	CO
324	91	61	56	49	0	0	19.3	1995	Longmont	Boulder	CO
338	66	59	56	47	0	0	18.6	1996	Longmont	Boulder	CO
191	44	41	34	34	0	0	18	1997	Longmont	Boulder	CO

No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
103	50	38	37	33	0	0	18.6	1998	Longmont	Boulder	CO
4	35	24	20	16	0	0	23.8	1994	Boulder	Boulder	CO
58	51	45	43	41	0	0	19.5	1995	Boulder	Boulder	CO
53	39	35	31	30	0	0	19.6	1996	Boulder	Boulder	CO
55	43	42	34	32	0	0	20.9	1997	Boulder	Boulder	CO
98	47	45	44	42	0	0	24.2	1998	Boulder	Boulder	CO
16	30	29	23	22	0	0	16.4	1998		Boulder	CO
109	100	86	56	56	0	0	27.8	1993	Delta	Delta	CO
127	77	70	66	64	0	0	29.5	1993	Delta	Delta	CO
329	148	105	105	105	0	0	31.5	1994	Delta	Delta	CO
342	70	69	63	63	0	0	24.4	1995	Delta	Delta	CO
340	71	67	63	60	0	0	25.6	1996	Delta	Delta	CO
202	104	55	50	50	0	0	23.1	1997	Delta	Delta	CO
50	64	40	39	38	0	0	22.8	1998	Delta	Delta	CO
46	27	24	24	23	0	0	15.9	1997		Delta	CO
59	45	35	35	32	0	0	17.6	1998		Delta	CO
8	59	28	23	20	0	0	24.4	1996		Delta	CO
51	90	78	65	53	0	0	26.9	1997		Delta	CO
53	77	68	64	46	0	0	24.8	1998		Delta	CO
72	109	101	87	87	0	0	38.9	1993	Denver	Denver	CO
83	102	89	77	69	0	0	33.1	1994	Denver	Denver	CO
59	52	50	48	44	0	0	27.9	1995	Denver	Denver	CO
56	59	54	44	43	0	0	28.1	1996	Denver	Denver	CO
89	67	66	64	62	0	0	26.4	1997	Denver	Denver	CO
53	48	47	44	43	0	0	26.7	1998	Denver	Denver	CO
60	111	103	93	91	0	0	40.5	1993	Denver	Denver	CO
57	96	73	65	63	0	0	34.9	1994	Denver	Denver	CO
57	57	57	49	46	0	0	28.7	1995	Denver	Denver	CO
59	58	50	44	43	0	0	28.3	1996	Denver	Denver	CO
59	66	66	64	62	0	0	26.3	1997	Denver	Denver	CO
52	60	51	49	49	0	0	28.2	1998	Denver	Denver	CO
343	162	122	112	108	1	1	31.8	1993	Denver	Denver	CO
342	110	104	99	88	0	0	28.3	1994	Denver	Denver	CO
337	75	65	56	53	0	0	21.1	1995	Denver	Denver	CO
338	74	67	57	56	0	0	20.4	1996	Denver	Denver	CO
242	86	71	70	67	0	0	23.1	1997	Denver	Denver	CO
361	108	81	79	74	0	0	30.9	1998	Denver	Denver	CO
58	111	110	103	82	0	0	38.8	1993	Denver	Denver	CO
58	82	70	69	61	0	0	31	1994	Denver	Denver	CO
60	44	42	40	40	0	0	25.2	1995	Denver	Denver	CO
60	56	53	53	49	0	0	27.8	1996	Denver	Denver	CO
58	92	91	84	62	0	0	28.5	1997	Denver	Denver	CO
58	73	66	59	51	0	0	28.9	1998	Denver	Denver	CO
62	117	111	104	84	0	0	39	1993	Denver	Denver	CO
57	79	71	68	64	0	0	32.6	1994	Denver	Denver	CO
59	57	45	44	41	0	0	26.9	1995	Denver	Denver	CO
61	63	53	51	48	0	0	27.7	1996	Denver	Denver	CO
59	94	93	89	62	0	0	28.9	1997	Denver	Denver	CO
55	71	69	54	47	0	0	27.1	1998	Denver	Denver	CO
336	161	119	106	100	1	1	29.4	1993	Denver	Denver	CO
335	74	72	72	71	0	0	25.4	1994	Denver	Denver	CO
350	91	80	56	50	0	0	21.4	1995	Denver	Denver	CO
345	81	70	66	66	0	0	22.8	1996	Denver	Denver	CO
348	68	66	61	60	0	0	21.8	1997	Denver	Denver	CO
300	77	75	71	69	0	0	29.5	1998	Denver	Denver	CO

No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
56	68	49	41	37	0	0	19	1993	Castle Rock	Douglas	CO
52	33	27	26	25	0	0	15.6	1994	Castle Rock	Douglas	CO
46	34	32	30	29	0	0	15.3	1995	Castle Rock	Douglas	CO
48	28	26	25	23	0	0	15.1	1996	Castle Rock	Douglas	CO
48	54	54	53	46	0	0	20.9	1997	Castle Rock	Douglas	CO
46	51	47	35	32	0	0	16.1	1998	Castle Rock	Douglas	CO
140	100	80	52	52	0	0	21	1993		Eagle	CO
130	43	38	38	36	0	0	16.7	1994		Eagle	CO
142	40	39	33	29	0	0	16.5	1995		Eagle	CO
99	77	52	43	39	0	0	17.5	1996		Eagle	CO
41	44	25	22	20	0	0	12.9	1997		Eagle	CO
43	94	46	27	20	0	0	15.8	1998		Eagle	CO
352	163	113	108	102	1	1	26.9	1993	Colorado Springs	El Paso	CO
112	50	47	43	42	0	0	20	1998	Colorado Springs	El Paso	CO
350	102	90	88	63	0	0	23	1994	Colorado Springs	El Paso	CO
349	84	72	69	65	0	0	21.1	1995	Colorado Springs	El Paso	CO
208	97	79	78	68	0	0	22.9	1997	Colorado Springs	El Paso	CO
353	93	76	76	72	0	0	21	1996	Colorado Springs	El Paso	CO
61	58	52	48	39	0	0	22.9	1993	Colorado Springs	El Paso	CO
61	58	52	39	36	0	0	19.6	1997	Colorado Springs	El Paso	CO
58	37	36	36	36	0	0	21.7	1998	Colorado Springs	El Paso	CO
61	28	28	27	27	0	0	18.3	1996	Colorado Springs	El Paso	CO
60	47	43	40	37	0	0	21.1	1994	Colorado Springs	El Paso	CO
54	30	29	29	28	0	0	18.7	1995	Colorado Springs	El Paso	CO
61	67	61	56	52	0	0	29.9	1993	Colorado Springs	El Paso	CO
59	59	55	47	46	0	0	24.8	1995	Colorado Springs	El Paso	CO
61	50	49	43	42	0	0	23.6	1997	Colorado Springs	El Paso	CO
61	47	47	41	41	0	0	24	1998	Colorado Springs	El Paso	CO
61	65	51	42	37	0	0	24.9	1996	Colorado Springs	El Paso	CO
60	87	63	51	50	0	0	29.2	1994	Colorado Springs	El Paso	CO
57	62	54	49	46	0	0	26.8	1995	Colorado Springs	El Paso	CO
61	67	47	42	40	0	0	26	1996	Colorado Springs	El Paso	CO
59	64	56	50	49	0	0	28.6	1994	Colorado Springs	El Paso	CO
60	67	59	53	51	0	0	29.2	1993	Colorado Springs	El Paso	CO
61	51	49	46	43	0	0	23.8	1997	Colorado Springs	El Paso	CO
60	47	46	44	43	0	0	25.5	1998	Colorado Springs	El Paso	CO
57	55	26	26	25	0	0	12.6	1993		El Paso	CO
59	42	27	26	25	0	0	12.3	1994		El Paso	CO
55	37	32	31	30	0	0	13.3	1995		El Paso	CO
59	32	31	27	26	0	0	12.1	1996		El Paso	CO
61	29	27	21	20	0	0	10.4	1997		El Paso	CO
59	32	26	25	25	0	0	12.5	1998		El Paso	CO
53	52	28	28	27	0	0	13.1	1993		El Paso	CO
57	44	28	26	25	0	0	12.3	1994		El Paso	CO
59	45	32	30	26	0	0	13.6	1995		El Paso	CO
60	48	29	27	26	0	0	12.6	1996		El Paso	CO
60	28	26	19	19	0	0	9.7	1997		El Paso	CO
55	35	30	25	24	0	0	12.8	1998		El Paso	CO
55	32	31	28	27	0	0	15.9	1993	Colorado Springs	El Paso	CO
54	33	30	29	27	0	0	16.6	1994	Colorado Springs	El Paso	CO
42	32	23	22	21	0	0	13.7	1995	Colorado Springs	El Paso	CO
49	34	29	29	28	0	0	15.5	1996	Colorado Springs	El Paso	CO
51	30	27	26	25	0	0	14.7	1997	Colorado Springs	El Paso	CO
56	36	31	29	27	0	0	16.7	1998	Colorado Springs	El Paso	CO
58	33	33	30	29	0	0	17.2	1993	Colorado Springs	El Paso	CO

No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
47	44	35	31	31	0	0	17.1	1994	Colorado Springs	El Paso	CO
45	32	25	23	22	0	0	13.8	1995	Colorado Springs	El Paso	CO
48	38	35	32	29	0	0	17.1	1996	Colorado Springs	El Paso	CO
54	30	29	28	26	0	0	15.2	1997	Colorado Springs	El Paso	CO
53	43	36	32	31	0	0	17.6	1998	Colorado Springs	El Paso	CO
58	52	40	38	37	0	0	22.6	1993		El Paso	CO
60	48	47	46	46	0	0	23.5	1994		El Paso	CO
60	52	48	46	41	0	0	22.9	1995		El Paso	CO
30	66	50	42	39	0	0	27.3	1996		El Paso	CO
60	60	36	33	33	0	0	15.8	1993		El Paso	CO
59	54	46	45	39	0	0	16.8	1994		El Paso	CO
56	32	29	28	25	0	0	12.4	1995		El Paso	CO
30	34	31	27	24	0	0	15.3	1996		El Paso	CO
54	40	37	33	29	0	0	15	1993		El Paso	CO
55	92	64	58	56	0	0	18.8	1994		El Paso	CO
59	63	56	41	39	0	0	18.2	1995		El Paso	CO
26	33	31	29	28	0	0	17.8	1996		El Paso	CO
48	78	56	53	49	0	0	30.8	1993	Colorado Springs	El Paso	CO
54	49	48	46	45	0	0	25.9	1994	Colorado Springs	El Paso	CO
49	72	57	43	41	0	0	25.2	1995	Colorado Springs	El Paso	CO
52	62	58	52	51	0	0	25.4	1996	Colorado Springs	El Paso	CO
55	42	42	41	39	0	0	22.3	1997	Colorado Springs	El Paso	CO
53	47	44	42	41	0	0	23.9	1998	Colorado Springs	El Paso	CO
339	64	61	55	53	0	0	22.9	1995	Colorado Springs	El Paso	CO
10	49	43	39	32	0	0	29	1994	Colorado Springs	El Paso	CO
339	64	61	55	53	0	0	22.9	1995	Colorado Springs	El Paso	CO
341	74	65	65	63	0	0	23.2	1996	Colorado Springs	El Paso	CO
177	48	47	44	42	0	0	21.6	1998	Colorado Springs	El Paso	CO
53	84	76	52	51	0	0	30.2	1993	Colorado Springs	El Paso	CO
57	82	53	52	49	0	0	28.1	1994	Colorado Springs	El Paso	CO
56	54	50	49	47	0	0	26.6	1995	Colorado Springs	El Paso	CO
30	70	48	37	36	0	0	27.5	1996	Colorado Springs	El Paso	CO
56	94	75	67	62	0	0	27.7	1993	Colorado Springs	El Paso	CO
54	55	50	45	45	0	0	23.6	1994	Colorado Springs	El Paso	CO
55	40	39	35	32	0	0	20	1995	Colorado Springs	El Paso	CO
54	80	49	45	42	0	0	22.2	1996	Colorado Springs	El Paso	CO
54	79	56	54	52	0	0	22.5	1997	Colorado Springs	El Paso	CO
57	37	37	36	34	0	0	20.2	1998	Colorado Springs	El Paso	CO
239	65	63	60	54	0	0	19.2	1995	Colorado Springs	El Paso	CO
137	57	55	51	46	0	0	21.5	1994	Colorado Springs	El Paso	CO
239	65	63	60	54	0	0	19.2	1995	Colorado Springs	El Paso	CO
337	84	72	65	65	0	0	20.6	1996	Colorado Springs	El Paso	CO
182	90	72	62	46	0	0	19.2	1998	Colorado Springs	El Paso	CO
52	82	58	52	51	0	0	31.1	1997	Colorado Springs	El Paso	CO
52	51	46	45	41	0	0	22.5	1997	Colorado Springs	El Paso	CO
56	39	36	35	31	0	0	18.7	1998	Coorado Springs	El Paso	CO
320	77	65	63	58	0	0	19.4	1993	Canon City	Fremont	CO
332	78	75	61	61	0	0	20.3	1994	Canon City	Fremont	CO
290	65	64	52	51	0	0	17.6	1995	Canon City	Fremont	CO
46	46	37	32	30	0	0	16.9	1996	Canon City	Fremont	CO
55	41	37	34	33	0	0	16.2	1997	Canon City	Fremont	CO
58	73	41	35	32	0	0	16.3	1998	Canon City	Fremont	CO
50	136	112	89	74	0	0	40.5	1993	Rifle	Garfield	CO
57	88	82	71	63	0	0	34.9	1994	Rifle	Garfield	CO
42	73	72	60	59	0	0	32.3	1995	Rifle	Garfield	CO

No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
46	97	78	75	65	0	0	32.7	1996	Rifle	Garfield	CO
37	65	63	53	49	0	0	29.5	1997	Rifle	Garfield	CO
59	70	57	52	42	0	0	24	1998	Rifle	Garfield	CO
51	108	82	72	56	0	0	24.6	1993	Glenwood Springs	Garfield	CO
43	58	55	49	32	0	0	22.1	1994	Glenwood Springs	Garfield	CO
56	69	66	51	44	0	0	22.4	1995	Glenwood Springs	Garfield	CO
52	66	40	35	33	0	0	19	1996	Glenwood Springs	Garfield	CO
54	45	36	32	29	0	0	16.9	1997	Glenwood Springs	Garfield	CO
47	72	65	40	39	0	0	20.3	1998	Glenwood Springs	Garfield	CO
175	97	91	91	85	0	0	31.9	1993		Gunnison	CO
168	100	96	93	91	0	0	32.2	1994		Gunnison	CO
138	116	96	91	91	0	0	31.6	1995		Gunnison	CO
60	103	82	82	63	0	0	29.6	1996		Gunnison	CO
60	110	80	79	70	0	0	34.6	1997		Gunnison	CO
114	137	109	74	71	0	0	29	1998		Gunnison	CO
24	141	91	87	76	0	0	46.7	1996		Gunnison	CO
217	228	215	203	177	4	9	51.4	1997		Gunnison	CO
323	207	149	145	142	1	1	37.9	1998		Gunnison	CO
50	76	69	61	55	0	0	27.3	1993	Arvada	Jefferson	CO
35	58	47	45	42	0	0	23.1	1994	Arvada	Jefferson	CO
60	41	36	35	34	0	0	18.2	1995	Arvada	Jefferson	CO
60	56	38	36	35	0	0	19.5	1996	Arvada	Jefferson	CO
53	70	70	64	53	0	0	21.3	1997	Arvada	Jefferson	CO
56	47	46	40	39	0	0	23.4	1998	Arvada	Jefferson	CO
58	52	40	32	26	0	0	14.3	1993		Jefferson	CO
59	24	22	20	20	0	0	12.7	1994		Jefferson	CO
57	31	25	22	21	0	0	9.7	1995		Jefferson	CO
60	31	30	28	23	0	0	11.7	1996		Jefferson	CO
58	25	23	22	18	0	0	9.4	1997		Jefferson	CO
59	37	31	24	23	0	0	12.6	1998		Jefferson	CO
61	62	45	36	30	0	0	15.1	1993		Jefferson	CO
55	26	25	23	23	0	0	13.9	1994		Jefferson	CO
59	34	26	24	24	0	0	11.5	1995		Jefferson	CO
60	32	29	28	28	0	0	13	1996		Jefferson	CO
58	25	23	22	19	0	0	10.7	1997		Jefferson	CO
61	37	32	25	24	0	0	13.9	1998		Jefferson	CO
59	62	47	34	31	0	0	15.1	1993		Jefferson	CO
59	27	25	23	23	0	0	14	1994		Jefferson	CO
57	35	26	22	22	0	0	11.3	1995		Jefferson	CO
61	33	28	28	28	0	0	13.1	1996		Jefferson	CO
60	26	22	22	19	0	0	11	1997		Jefferson	CO
61	36	32	25	24	0	0	14.1	1998		Jefferson	CO
58	67	48	35	32	0	0	15.6	1993		Jefferson	CO
58	27	27	26	26	0	0	14.3	1994		Jefferson	CO
54	87	57	46	39	0	0	16.6	1995		Jefferson	CO
59	32	28	26	26	0	0	13.1	1996		Jefferson	CO
61	25	24	21	20	0	0	10.6	1997		Jefferson	CO
59	37	32	30	25	0	0	14.3	1998		Jefferson	CO
55	34	26	25	21	0	0	11	1995		Jefferson	CO
56	36	29	28	25	0	0	13.7	1996		Jefferson	CO
59	23	20	19	18	0	0	10.1	1997		Jefferson	CO
60	37	30	25	25	0	0	13.1	1998		Jefferson	CO
57	37	31	28	25	0	0	12.3	1995		Jefferson	CO
60	41	39	33	32	0	0	14.7	1996		Jefferson	CO
57	26	23	21	21	0	0	11.3	1997		Jefferson	CO

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No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
56	38	31	28	27	0	0	14.8	1998		Jefferson	CO
55	104	68	51	47	0	0	24.4	1993	Golden	Jefferson	CO
34	55	53	41	34	0	0	20.7	1994	Golden	Jefferson	CO
56	38	37	35	30	0	0	15.9	1995	Golden	Jefferson	CO
56	43	31	30	26	0	0	16	1996	Golden	Jefferson	CO
6	33	28	20	19	0	0	23.5	1997	Golden	Jefferson	CO
42	93	92	86	75	0	0	42.2	1996	Durango	La Plata	CO
163	118	106	97	96	0	0	38.4	1997	Durango	La Plata	CO
254	206	77	76	71	1	1	30.2	1998	Durango	La Plata	CO
37	39	35	26	24	0	0	16.2	1997	Durango	La Plata	CO
179	83	73	59	47	0	0	17.9	1998	Durango	La Plata	CO
61	71	57	57	44	0	0	23.5	1993	Durango	La Plata	CO
46	37	32	31	29	0	0	17.2	1994	Durango	La Plata	CO
51	41	40	33	32	0	0	17.4	1995	Durango	La Plata	CO
58	57	55	47	39	0	0	18.3	1996	Durango	La Plata	CO
160	54	45	44	43	0	0	17.9	1997	Durango	La Plata	CO
168	94	57	44	37	0	0	17.5	1998	Durango	La Plata	CO
56	62	54	49	42	0	0	22.4	1993	Fort llins	Larimer	CO
72	51	45	42	41	0	0	21.6	1994	Fort llins	Larimer	CO
52	57	47	45	44	0	0	22.3	1995	Fort llins	Larimer	CO
51	61	52	38	35	0	0	20.4	1996	Fort llins	Larimer	CO
60	40	34	34	32	0	0	15.7	1997	Fort llins	Larimer	CO
102	34	32	32	28	0	0	16.2	1998	Fort llins	Larimer	CO
90	74	53	49	40	0	0	21	1993		Larimer	CO
34	50	39	38	37	0	0	23.2	1994		Larimer	CO
58	51	35	32	31	0	0	21.1	1993	Fruita	Mesa	CO
57	43	42	41	39	0	0	22.1	1994	Fruita	Mesa	CO
43	35	34	31	30	0	0	20	1995	Fruita	Mesa	CO
44	36	36	33	33	0	0	17.7	1996	Fruita	Mesa	CO
55	36	36	32	30	0	0	18.4	1997	Fruita	Mesa	CO
175	67	62	61	56	0	0	25	1993	Grand Junction	Mesa	CO
171	63	54	50	50	0	0	24.3	1994	Grand Junction	Mesa	CO
148	56	46	43	42	0	0	22.3	1995	Grand Junction	Mesa	CO
166	64	63	49	44	0	0	21.9	1996	Grand Junction	Mesa	CO
113	50	48	48	46	0	0	22	1997	Grand Junction	Mesa	CO
45	51	44	41	39	0	0	22.6	1998	Grand Junction	Mesa	CO
356	60	56	55	49	0	0	21.5	1993	Grand Junction	Mesa	CO
364	55	54	54	54	0	0	21.4	1994	Grand Junction	Mesa	CO
347	49	48	46	46	0	0	21.8	1995	Grand Junction	Mesa	CO
359	50	49	45	45	0	0	20.6	1996	Grand Junction	Mesa	CO
342	60	49	46	42	0	0	19.6	1997	Grand Junction	Mesa	CO
337	55	51	47	45	0	0	19.8	1998	Grand Junction	Mesa	CO
59	62	41	39	36	0	0	23.3	1993	Grand Junction	Mesa	CO
58	54	45	45	45	0	0	22.2	1994	Grand Junction	Mesa	CO
56	41	38	33	32	0	0	18.5	1995	Grand Junction	Mesa	CO
60	40	39	38	36	0	0	19.9	1996	Grand Junction	Mesa	CO
59	43	37	35	34	0	0	17.6	1997	Grand Junction	Mesa	CO
53	71	40	33	29	0	0	20.2	1998	Grand Junction	Mesa	CO
6	41	32	31	31	0	0	27.3	1995	Montrose	Montrose	CO
58	66	60	58	52	0	0	26.7	1996	Montrose	Montrose	CO
61	65	55	48	47	0	0	24.9	1997	Montrose	Montrose	CO
38	50	49	47	46	0	0	24.8	1998	Montrose	Montrose	CO
7	81	54	52	42	0	0	41.7	1997		Montrose	CO
113	79	79	74	71	0	0	35.1	1998		Montrose	CO
348	98	88	84	82	0	0	23.9	1993	Aspen	Pitkin	CO

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No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
329	88	76	75	66	0	0	22.1	1994	Aspen	Pitkin	CO
334	86	83	75	74	0	0	23.3	1995	Aspen	Pitkin	CO
331	88	66	51	51	0	0	19.4	1996	Aspen	Pitkin	CO
334	92	89	74	68	0	0	21	1997	Aspen	Pitkin	CO
340	68	64	58	56	0	0	20	1998	Aspen	Pitkin	CO
282	62	61	61	53	0	0	22.6	1998	Aspen	Pitkin	CO
89	67	66	60	60	0	0	18.3	1993	Aspen	Pitkin	CO
53	81	45	43	40	0	0	19.5	1994	Aspen	Pitkin	CO
180	77	71	70	65	0	0	23.4	1993	Lamar	Prowers	CO
156	142	112	105	90	0	0	24.9	1994	Lamar	Prowers	CO
180	132	87	77	71	0	0	24.7	1995	Lamar	Prowers	CO
340	126	80	73	70	0	0	24.3	1996	Lamar	Prowers	CO
332	101	92	88	66	0	0	23	1997	Lamar	Prowers	CO
351	137	100	98	82	0	0	26.4	1998	Lamar	Prowers	CO
360	54	54	53	47	0	0	20.8	1993	Lamar	Prowers	CO
348	79	79	73	67	0	0	22	1994	Lamar	Prowers	CO
331	147	93	88	86	0	0	22.3	1995	Lamar	Prowers	CO
243	145	65	54	54	0	0	18.3	1996	Lamar	Prowers	CO
312	110	98	55	54	0	0	17.5	1997	Lamar	Prowers	CO
323	89	86	76	63	0	0	21.4	1998	Lamar	Prowers	CO
54	52	51	43	43	0	0	26.1	1993	Pueblo	Pueblo	CO
54	63	54	53	50	0	0	29.6	1994	Pueblo	Pueblo	CO
51	100	86	56	54	0	0	26.2	1995	Pueblo	Pueblo	CO
52	59	49	48	47	0	0	25.8	1996	Pueblo	Pueblo	CO
57	88	56	56	43	0	0	26.8	1997	Pueblo	Pueblo	CO
31	51	37	33	33	0	0	21.7	1998	Pueblo	Pueblo	CO
53	60	52	49	45	0	0	24.8	1998	Pueblo	Pueblo	CO
352	158	151	139	128	1	1	32.7	1993	Steamboat Springs	Routt	CO
342	154	148	136	130	0	0	31.8	1994	Steamboat Springs	Routt	CO
343	139	135	131	123	0	0	31.7	1995	Steamboat Springs	Routt	CO
307	158	137	134	125	1	1	31.5	1996	Steamboat Springs	Routt	CO
339	117	112	99	99	0	0	28	1997	Steamboat Springs	Routt	CO
352	82	77	75	75	0	0	25.7	1998	Steamboat Springs	Routt	CO
61	109	105	97	93	0	0	29.7	1996	Steamboat Springs	Routt	CO
116	91	86	84	79	0	0	27.8	1997	Steamboat Springs	Routt	CO
168	128	126	106	96	0	0	28	1993	Steamboat Springs	Routt	CO
153	142	124	121	118	0	0	28.2	1994	Steamboat Springs	Routt	CO
145	118	114	103	97	0	0	23	1995	Steamboat Springs	Routt	CO
74	83	77	54	54	0	0	23.2	1996	Steamboat Springs	Routt	CO
330	135	126	118	117	0	0	39.4	1993		San Miguel	CO
281	153	127	123	108	0	0	33.8	1994		San Miguel	CO
273	119	103	95	90	0	0	34.8	1995		San Miguel	CO
321	107	105	101	89	0	0	25.8	1996		San Miguel	CO
297	96	80	75	74	0	0	24.9	1997		San Miguel	CO
316	70	65	65	63	0	0	23.9	1998		San Miguel	CO
19	27	24	24	22	0	0	16.3	1996		San Miguel	CO
272	82	76	75	69	0	0	26.4	1997		San Miguel	CO
362	90	72	58	57	0	0	25.5	1998		San Miguel	CO
47	44	42	41	39	0	0	17	1995		San Miguel	CO
52	130	95	92	83	0	0	24.4	1993		Summit	CO
43	126	90	84	73	0	0	24.1	1994		Summit	CO
47	97	68	52	47	0	0	18	1995		Summit	CO
40	50	26	26	23	0	0	13.4	1996		Summit	CO
58	95	75	37	32	0	0	17.1	1997		Summit	CO
110	125	69	67	65	0	0	19.2	1998		Summit	CO

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No. of 24-hr Values	1st Max of 24-hr Value	2nd Max of 24-hr Value	3rd Max of 24-hr Value	4th Max of 24 hr Value	Actual # of Exceedences	Est. # of Exceedences	Annual Mean	Year	City	County	State
11	67	61	44	43	0	0	34.5	1993		Summit	CO
42	82	62	59	53	0	0	27.4	1994		Summit	CO
16	76	72	47	43	0	0	32.4	1995		Summit	CO
48	78	56	49	40	0	0	24.9	1996		Summit	CO
52	62	40	38	38	0	0	18.8	1997		Summit	CO
50	47	46	44	44	0	0	21.9	1998		Summit	CO
12	139	122	83	54	0	0	57.2	1994		Teller	CO
96	306	266	214	204	6	19	51.5	1995		Teller	CO
316	235	195	158	157	4	5	39.1	1996		Teller	CO
228	135	121	120	111	0	0	39.9	1997		Teller	CO
249	139	124	120	109	0	0	41	1998		Teller	CO
150	120	99	80	76	0	0	22.6	1993	Greeley	Weld	CO
143	75	57	56	48	0	0	23.1	1994	Greeley	Weld	CO
132	60	59	51	46	0	0	19.9	1995	Greeley	Weld	CO
159	60	56	45	42	0	0	17.7	1996	Greeley	Weld	CO
114	133	56	52	46	0	0	17.8	1997	Greeley	Weld	CO
107	40	39	36	32	0	0	16.5	1998	Greeley	Weld	CO
50	110	82	73	70	0	0	30.5	1993		Weld	CO
56	89	68	53	49	0	0	27.5	1994		Weld	CO
23	53	45	39	36	0	0	21	1995		Weld	CO

*Colorado Air Quality Monitors for Particulate Matter (All Years)

* Monitor Values In Micrograms Per Cubic Meter of Air (ug/m3)

Appendix G – RESRAD Results for the Resident Rancher Scenario.

The RSAL working group has committed to model the Resident Rancher scenario (both adult and child cases) as described in the Risk Assessment Corporation (RAC) Independent Calculation using RESRAD 6.0, for the purpose of comparing the computational methods employed by RAC to those employed by the work group. On the surface, this task appears to be straightforward – simply input the RAC parameters described in Tasks 3 and 5, (RAC, 1999, and RAC, 2000) into RESRAD 6.0 and perform the computation.

However the Working Group soon learned that is was not a simple matter to duplicate the inputs that RAC used for annual average air mass loading (dust in air). For the Independent Calculation, RAC computed this parameter not as a distribution, but as a series of calculations, which are combined, with other parameters selected from distributions. Moreover, the calculated values of mass loading which RAC created were heavily influenced by the assumptions of pre clean-up conditions, placement of the receptor at a point of maximum air concentration, and inclusion of probabilistic impacts of a fire. RAC's calculation of the mass loading parameter (for each realization) is performed by a RAC developed code that is beyond the scope of the RSAL Working Group to reproduce. With this in mind, the Working Group has sought to formulate a value for the mass loading input parameter that is consistent with RAC's work.

The Working Group used the Perl-script code developed by RAC (RAC, 2000, Appendix A) to produce a distribution of intermediate values of annual average mass loading. (These are the values of mass loading that RAC input into their copy of the RESRAD code, along with samples from each of their various distributions of other physical parameters, for each realization.) From the distribution of 1000 values of mass loading calculated by the RAC algorithm, the 90th and 95th percentile values were selected. The work group then selected conservative single-point estimates for the other distributed parameters, which RAC used, and calculated RSALs for plutonium and americium for the case of the adult and child resident rancher using single deterministic runs of RESRAD 6.0. Although this conservatively approximates the RAC approach, it does not duplicate it. In order to do so one would have to use the entire RAC code for selecting samples of each parameter distribution every time the mass loading value is computed. RAC's Independent Calculation has already done this. The approximation described above, serves as a benchmark or point of comparison of the working group's computer model with RAC's total program for this scenario.

Modeling Assumptions:

All active pathways and all input parameters for the resident rancher scenario are identical to those found in the RAC Task 3 Report (RAC, 1999) except for substitutions of fixed values for uptake parameters and distribution coefficients, and the use of two fixed values of mass loading taken from a distribution of RAC calculated values. All features of the rancher scenario are the same as modeled by RAC. All exposure pathways except aquatic food and radon are active in this calculation. Consistent with the RAC calculation, the contaminated fractions of drinking water, irrigation water and livestock water are all set to zero values (RAC, 2000, Appendix A).

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1. The area of the contaminated zone is a 10 million square meter area that is uniformly contaminated to the RSAL concentration. The resident is located in the center. This is a conservative substitution, which is consistent with RESRAD input requirements.

2. Both dose limits of 25 mrem per year and 15 mrem per year are modeled. This permits easy comparison to other calculations in this task and to RAC's calculation. These computations use the same dose conversion factors for adults and children as used by RAC (plutonium type "S" absorption, child DCFs for age 10), unlike the remainder of calculations in this task (plutonium type "M", child DCF's for age 1, which are more conservative).

3. RESRAD single default values of the distribution coefficients and plant, meat, and milk uptake fractions for plutonium and americium are used in lieu of the distributions used by RAC. The fixed default values in RESRAD lie on the conservative side of RAC's distributions, and have little impact on the results which are dominated by the impact of high values of mass loading for inhalation.

4. The rancher adult and child spend all of their time on the site, with times outdoors of 40% and 25%, respectively. Indoor dust and gamma shielding factors are the same as used by RAC .

5. Breathing rates, and consumption rates of homegrown produce, meat, milk and drinking water (from shallow groundwater) are the same values as described in RAC Task 3.

6. Single values for annual average mass loading for inhalation/foliar deposition (3,180 and 8,920 micrograms per cubic meter for the 90th and 95th percentile, respectively) are used. These are derived by using the RAC mass loading subroutine to calculate a distribution of 1000 points, followed by selection of the 90th and 95th percentile values of this distribution.

7. The sum of ratios method described elsewhere in this Task is applied to the single radionuclide soil guidelines calculated for plutonium and for americium by RESRAD 6.0. The assumption is made that americium is present at 15.3 % of the plutonium soil concentration across the entire site, based upon the best data available from the 903 Pad studies. These results are consistent with weapons grade plutonium that has aged between 35 and 45 years.

Results and Discussion:
Tables G-1 and G-2 summarize the values of RSALs calculated by the sum of ratios method for the 90th and the 95th percentile values of RAC calculated annual average mass loading of one micron particles, respectively. The high values of mass loading clearly drive the dose calculation. At the 90th percentile the combination of inhalation and plant ingestion dose (which is strongly controlled by deposition of dust on plants) account for approximately 85% of the total dose. For the 95th percentile, this same combination accounts for up to 95% of the total dose.

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Table G-1. RSALs (pCi/g) for resident rancher at 90th percentile value of RAC-calculated mass loading (3180 ug/m3). Inhalation pathway contributions range from 64-70% of total dose. For comparative purposes only.

RSAL Isotope	**Adult 25 mrem/yr**	**Child (10) 25 mrem/yr**	**Adult 15 mrem/yr**	**Child (10) 15 mrem/yr**
Pu RSAL	45	49	27 ***	30
Am RSAL	7	8	4	5

*** most comparable RSAL value to RAC Task 5 Report value.

Table G-2 RSALs (pCi/g) for Resident Rancher at 95th percentile value of RAC calculated mass loading (8920 ug/m3). Inhalation pathway contributions range from 81-85% of total dose. For comparative purposes only.

RSAL Isotope	**Adult 25 mrem/yr**	**Child (10) 25 mrem/yr**	**Adult 15 mrem/yr**	**Child (10) 15 mrem/yr**
Pu RSAL	20	22	12	13
Am RSAL	3	3	2	2

More than one third of the annual average mass loading values calculated by RAC's subroutine exceed the highest actual measured value for PM10 annual averages reported to the Aerometric Information Retrieval System, or AIRS (268 ug/m3 in Mexicali, Baja California in 2000) and greatly exceed the National Ambient Air Quality Standard for PM10 annual average (50 ug/m3). Specifically, the 90th and 95th percentile values of RAC's distribution, used in this calculation are 12 and 33 times higher, respectively than the highest PM10 averages reported to AIRS to date.

The most comparable RSAL value for the RESRAD 6.0 resident rancher scenario to that calculated in RAC's Task 5 Report, is the adult value for a 15 mrem/yr dose limit at the 90th percentile of RAC's mass loading distribution. As can be seen from Table IV-3, the Working Group's value of 27 pCi/g for Pu agrees rather well with RAC's 35 pCi/g. This agreement reconfirms that differences between the Working Group's dose based RSAL values and RAC's are largely due to differences in choice of input parameters and dose conversion factors and cannot be attributed to differences in computer models.

Table G-3 following is a complete listing of the RESRAD 6.0 parameters that were used in the adult and child resident rancher calculations.

Table G-3

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 Input VALUE	Resident Rancher (ADULT)	Resident Rancher (CHILD)
Pathways					
External Gamma		active	active	active	active
Inhalation		active	active	active	active
Plant Ingestion		active	active	active	active
Meat Ingestion		active	Suppressed	active	active
Milk Ingestion		active	Suppressed	active	active
Aquatic Foods		active	Suppressed	Suppressed	Suppressed
Drinking Water		active	Suppressed	active	active
Soil Ingestion		active	active	active	active
Radon		active	Suppressed	Suppressed	Suppressed
Initial Principal Radionuclide					
Activity in Contaminated Zone	pCi/g		Am-241	0.111	0.111
	pCi/g		Pu-238	0.0132	0.0132
	pCi/g		Pu-239	0.843	0.843
	pCi/g		Pu-240	0.157	0.157
	pCi/g		Pu-241	0.798	0.798
	pCi/g		Pu-242	7.62E-06	7.62E-06
Basic Radiation Dose Limit	mrem/y	25	15	15 & 25	15 & 25
Time for Calculations	y	1	0.2	29	29
Time for Calculations	y	3	1	1029	1029
Time for Calculations	y	10	5	not used	not used
Time for Calculations	y	30	not used	not used	not used
Time for Calculations	y	100	not used	not used	not used
Time for Calculations	y	300	not used	not used	not used
Time for Calculations	y	1000	not used	not used	not used
Time for Calculations	y	not used	not used	not used	not used
Time for Calculations	y	not used	not used	not used	not used

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Occupancy, Inhalation, and External Gamma					
Inhalation Rate	m3/y	8400	7000	10800	8600
Mass Loading for Inhalation	g/m3	0.0001	0.000026	0.00318(90%) & 0.008920(95%)	0.00318(90%) & 0.008920(95%)
Exposure Duration	y	30	30	30 not used	30 not used
Indoor Dust Filtration Factor		0.4	1	0.7	0.7
External Gamma Shielding Factor		0.7	0.8	0.7	0.7
Indoor Time Fraction		0.5	1	0.6	0.75
Outdoor Time Fraction		0.25	0	0.4	0.25
Shape Factor for external gamma		1	1	1	1
Area of Contaminated Zone	m2	10000	40000	10,000,000	10,000,000
Thickness of Contaminated Zone	m	2	0.15	0.2	0.2
Length Parallel to Aquifer Flow	m	100	200	3000	3000

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 VALUE	Resident Rancher (ADULT)	Resident Rancher (CHILD)
Cover and Contaminated Zone Hydrological Data					
Cover Depth	m	0	not used	No Cover	No Cover
Density of Cover Material	g/cm3	1.5	not used	No Cover	No Cover
Cover Erosion Rate	m/y	0.001	not used	No Cover	No Cover
Density of Contaminated Zone	g/cm3	1.5	1.8	1.8	1.8
Contaminated Zone Erosion Rate	m/y	0.001	0.0000749	0.0000749	0.0000749
Contaminated Zone Total Porosity		0.4	0.3	0.3	0.3
Contaminated Zone Field Capacity		0.2	0.1	0.1	0.1
Contaminated Zone Hydraulic Conductivity	m/y	10	44.5	44.5	44.5
Contaminated Zone b Parameter		5.3	10.4	10.4	10.4
Humidity in Air	g/m3	8	not used	**not used**	**not used**
Evapotranspiration Coefficient		0.5	0.253	0.92	0.92
Average Annual Wind Speed	m/s	2	2	4.2	4.2
Precipitation	m/y	1	0.381	0.381	0.381
Irrigation	m/y	0.2	1	0	0
Irrigation Mode		overhead	overhead	overhead	overhead
Runoff Coefficient		0.2	0.004	0.2	0.2

Watershed Area	m2	1000000	8280000	8280000	8280000
Accuracy for Water/Soil Computations		0.001	0.001	0.001	0.001
Uncontaminated Unsaturated Zone Parameters					
Number of Unsaturated Zone Strata		1	1	1	1
Thickness	m	4	3	3	3
Density	g/cm3	1.5	1.8	1.8	1.8
Total Porosity		0.4	0.3	0.3	0.3
Effective Porosity		0.2	0.1	0.1	0.1
Field Capacity		0.2	0.1	0.1	0.1
Hydraulic Conductivity	m/y	10	44.5	44.5	44.5
b Parameter		5.3	10.4	10.4	10.4
Radionuclide Transport Factors					
Distribution Coefficient Contaminated Zone	cm3/g	-		Pu = 2000, Am = 20	Pu = 2000, Am = 20
Distribution Coefficient Unsaturated Zone	cm3/g	-		Pu = 2000, Am = 20	Pu = 2000, Am = 20
Distribution Coefficient Saturated Zone	cm3/g	-		Pu = 2000, Am = 20	Pu = 2000, Am = 20
Time since placement of materials	year	0		0	0
Solubility Limit	mol/l	0		0	0
Leach Rate	year-1	0		0	0

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 VALUE	Resident Rancher (ADULT)	Resident Rancher (CHILD)
Saturated Zone Hydrological Data					
Density of Saturated Zone	g/cm3	1.5	1.8	1.8	1.8
Saturated Zone Total Porosity		0.4	0.3	0.3	0.3

Saturated Zone Effective Porosity		0.2	**0.1**	0.1	0.1
Saturated Zone Field Capacity		0.2	**0.1**	0.1	0.1
Saturated Zone Hydraulic Conductivity	m/y	100	**44.5**	44.5	44.5
Saturated Zone Hydraulic Gradient		0.02	**0.15**	0.15	0.15
Saturated Zone b Parameter		5.3	**not used**	5.3	5.3
Water Table Drop Rate		0.001	**0**	0	0
Well Pump Intake Depth (below water table)	m	10	10	10	10
Model: nondispersion (ND) or Mass-Balance (MB)		ND	ND	ND	ND
Well Pumping Rate	m3/y	250	250	250	250
Ingestion Pathway, Dietary Data					
Fruit, Vegetable and Grain Consumption	kg/y	160	**40.1**	190	240
Leafy Vegetable Consumption	kg/y	14	**2.6**	64	42
Milk Consumption	l/y	92	**not used**	110	200
Meat and Poultry Consumption	kg/y	63	**not used**	95	60
Fish Consumption	kg/y	5.4	**not used**	**not used**	**not used**
Other Seafood Consumption	kg/y	0.9	**not used**	**not used**	**not used**
Soil Ingestion	g/y	36.5	**70**	75	75
Drinking Water Intake	l/y	510	**not used**	730	550
Contaminated Fraction, Drinking Water		1	**not used**	0	0
Contaminated Fraction, Household Water		1	**not used**	**not used**	**not used**
Contaminated Fraction, Livestock Water		1	**not used**	0	0
Contaminated Fraction, Irrigation Water		1	**0**	0	0
Contaminated Fraction, Aquatic Food		0.5	**not used**	**not used**	**not used**
Contaminated Fraction, Plant Food		-1	**1**	1	1
Contaminated Fraction, Meat		-1	**not used**	1	1
Contaminated Fraction, Milk		-1	**not used**	1	1
Ingestion Pathway, Nondietary Data					
Livestock Fodder Intake For Meat	kg/day	68	**not used**	68	68
Livestock Fodder Intake for Milk	kg/day	55	**not used**	55	55
Livestock Water Intake For Meat	l/d	50	**not used**	0	0
Livestock Water Intake For Milk	l/d	160	**not used**	0	0
Livestock Intake For Soil	kg/day	0.5	**not used**	0.5	0.5
Mass Loading for Foliar Deposition	g/m3	0.0001	**0.0001**	0.00318(90%) & 0.00892(95%)	0.00318(90%) & 0.00892(95%)
Depth of Soil Mixing Layer	m	0.15	**0.15**	0.03	0.03
Depth of Roots	m	0.9	**0.9**	0.9	0.9
Groundwater Fractional Usage, Drinking Water		1	**not used**	1	1
Groundwater Fractional Usage, Household Water		1	**not used**	**not used**	**not used**

Groundwater Fractional Usage, Livestock Water		1	not used	1	1
Groundwater Fractional Usage, Irrigation Water		1	not used	1	1

RESRAD 6.0 INPUT PARAMETERS	UNITS	RESRAD 6.0 Default	1996 VALUE	Resident Rancher (ADULT)	Resident Rancher (CHILD)
Plant Factors					
Wet Weight Crop Yield, Non-Leafy	kg/m2	0.7	0.7	0.7	0.7
Length of Growing Season, Non-Leafy	years	0.17	0.17	0.17	0.17
Translocation Factor, Non-Leafy		0.1	0.1	0.1	0.1
Weathering Removal Constant	1/year	20	20	20	20
Wet Foliar Interception Fraction, Non-Leafy		0.25	0.25	0.25	0.25
Dry Foliar Interception Fraction, Non-Leafy		0.25	0.25	0.25	0.25
Wet Weight Crop Yield, Leafy	kg/m2	1.5	1.5	1.5	1.5
Length of Growing Season, Leafy	years	0.25	0.25	0.25	0.25
Translocation Factor, Leafy		1	1	1	1
Wet Foliar Interception Fraction, Leafy		0.25	0.25	0.25	0.25
Dry Foliar Interception Fraction, Leafy		0.25	0.25	0.25	0.25
Wet Weight Crop Yield, Fodder	kg/m2	1.1	1.1	1.1	1.1
Length of Growing Season, Fodder	years	0.08	0.08	0.08	0.08
Translocation Factor, Fodder		1	1	1	1
Weathering Removal Constant, Fodder	1/year	20	20	20	20
Wet Foliar Interception Fraction, Fodder		0.25	0.25	0.25	0.25
Dry Foliar Interception Fraction, Fodder		0.25	0.25	0.25	0.25
Storage Times Before Use Data					
Fruits, Non-Leafy Vegetables and Grain	days	14	14	14	14
Leafy Vegetables	days	1	1	1	1
Milk	days	1	1	1	1
Meat	days	20	20	20	20
Fish	days	7	7	not used	not used
Crustacea and Mollusks	days	7	7	not used	not used
Well Water	days	1	1	1	1
Surface Water	days	1	1	1	1
Livestock Fodder	days	45	45	45	45