NAC-0027_R1

# Dose Assessment for the Clive DU PA

## Clive DU PA Model v1.2

8 July 2014

Prepared by
NEPTUNE AND COMPANY, INC.
1505 15th St, Suite B, Los Alamos, NM 87544

| | | |
|---|---|---|
| 1. Title: Dose Assessment for the Clive DU PA | | |
| 2. Filename: Dose Assessment v1.2.docx | | |
| 3. Description: Documentation of the human exposure and dose assessment for the Clive DU PA Model v1.2 | | |
| | Name | Date |
| 4. Originator | R. Perona, R. Lee | 30 May 2014 |
| 5. Reviewer | P. Black | 31 May 2014 |
| 6. Remarks | | |

This page is intentionally blank, aside from this statement.

# CONTENTS

FIGURES ..... v
TABLES ..... vi
1.0 Summary of Input Parameter Values ..... 1
2.0 Purpose and Context ..... 9
3.0 Exposure-Dose Model Implementation ..... 9
3.1 Summary of Exposure-Dose Model Scope ..... 9
3.2 Exposure Scenarios ..... 11
3.2.1 Ranching ..... 12
3.2.2 Recreation ..... 13
3.2.3 Other Potential Receptors ..... 13
3.3 Assessment Endpoints ..... 14
3.3.1 Individual Dose ..... 14
3.3.2 As Low As Reasonably Achievable (ALARA) ..... 15
3.3.3 Collective Dose ..... 17
3.4 Modeling Doses ..... 18
3.4.1 Individual Doses ..... 18
3.4.2 Collective Dose ..... 20
3.4.3 Dose Conversion Factors ..... 20
3.4.4 Additional Sources of Uncertainty ..... 26
3.4.5 Non-Cancer Toxicity Endpoints ..... 26
4.0 Equations and Parameters of the Exposure-Dose Container ..... 27
4.1 Organization ..... 27
4.2 Environmental Concentrations ..... 28
4.3 Exposure Parameters ..... 29
4.4 DCFs ..... 30
4.5 PDCFs ..... 31
4.5.1 Inhalation PDCF Equations ..... 32
4.5.2 External PDCF Equations ..... 32
4.5.3 Ingestion PDCF Equations ..... 33
4.6 TEDE ..... 34
4.6.1 Inhalation TEDE Equations ..... 34
4.6.2 External Radiation TEDE Equations ..... 36
4.6.3 Ingestion TEDE Equations ..... 37
5.0 References ..... 40
Appendix I: Discussion of Derivations of Selected Parameter Distributions ..... 44

# FIGURES

Figure 1. Geometric mean of body weight as a function of age. ....45
Figure 2. Examples of distributions for body weight. ....46
Figure 3. Geometric means for ventilation rate, as a function of age and gender. ....47
Figure 4. Examples of ventilation rate distributions for different activities (20-year-old male). ....47
Figure 5. Distributions for soil ingestion, representing different tracers. ....48
Figure 6. Distributions for home-produced meat ingestion rates. ....49
Figure 7. Example distributions for sedentary plus sleeping time/day and sleeping time/day (30-year-old female). ....50
Figure 8. Distributions for light, medium, and heavy activity time/day (30-year-old female)....50
Figure 9. Distribution for the total number of individuals at the site during a given year. ....51
Figure 10. Distribution for the average day-trip time. ....52
Figure 11. Distribution for dust loading (overlaid on a histogram of simulated values). ....53
Figure 12. Distribution for Rancher exposure frequency. ....53
Figure 13. Distribution for Sport OHVer exposure frequency. ....54
Figure 14. Distribution for Hunter exposure frequency. ....55
Figure 15. Distribution for rest area caretaker exposure frequency. ....55
Figure 16. Distributions for meat loss (preparation and post-cooking). ....56
Figure 17. Distribution for the average cattle range acreage. ....57
Figure 18. Distribution for alpha particle REF. ....58
Figure 19. Distribution for electron and photon REFs. ....58

# TABLES

Table 1. Exposure dose input parameters summary .....1
Table 2. Exposure pathways summary .....11
Table 3. Beef transfer factors (Bq/kg per Bq/d) .....39

# 1.0 Summary of Input Parameter Values

Following is a brief summary of input values used parameters employed in the "exposure-dose" (ED) component of the Clive Performance Assessment (PA) model that is the subject of this white paper. Please see Appendix I in this document, the companion spreadsheet *Dose Assessment Appendix II*, and the *Model Parameters* white paper (Appendix 16) for further justifications of selected values, and the text for further explanation.

For distributions, the following notation is used:

- N( μ, σ, [*min*, *max*] ) represents a normal distribution with mean μ and standard deviation σ, and optional truncation at the specified *min*imum and *max*imum,
- LN( *GM*, *GSD*, [*min*, *max*] ) represents a log-normal distribution with geometric mean GM and geometric standard deviation GSD, and optional *min* and *max*,
- U( *min*, *max* ) represents a uniform distribution with lower bound *min* and upper bound *max*,
- Beta( μ, σ, *min*, *max* ) represents a generalized beta distribution with mean μ, standard deviation σ, minimum *min*, and maximum *max*,
- Gamma( μ, σ ) represents a gamma distribution with mean μ and standard deviation σ, and
- TRI( *min*, *m*, *max* ) represents a triangular distribution with lower bound *min*, mode *m*, and upper bound *max*.

**Table 1. Exposure dose input parameters summary**

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| **"Inner Loop" human exposure and dose factors; sampled multiple times within a realization** | | | | | | |
| Dose conversion factors (DCFs) | Sv/Bq; Sv-m$^3$ / Bq-s | Distributions for some DCFs are derived based upon Kocher et al, 2005 REFs (see below). See also Dose Assessment Appendix II.xls | | EPA, 1999; and others | | |
| Radiation effectiveness factors (REFs) | Unitless | Alpha:<br>LN( 1.81e+01, 2.37+00)<br><br>Photon < 30 keV:<br>LN( 2.45, 1.55 )<br><br>Photon 30-250 keV:<br>LN( 1.96, 1.48)<br><br>Electron:<br>LN( 2.41, 1.44) | | Kocher et al., 2005 | 14, 15; p. 26 | Particle- and energy-specific values. Based upon lognormal fits to percentiles presented in Kocher et al., 2005 |
| Uranium oral reference dose | mg/kg-day | Discrete(<br>0.5, 0.0006;<br>0.5, 0.003) | | EPA, 2011;<br>EPA, 2000 | | Equal probability assigned to Office of Water and Superfund criteria. |
| Age | yr | N( 25.7, 20.3 ), truncated at 16 and 60 | | USFS, 2005 | 2, p. 8 | |
| Gender | | Male: 60.8%<br>Female: 39.2% | | USFS, 2005 | 2, p. 8 | |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| Body weight | kg | Male:<br>LN( exp( 4.08+1.64e-2*Age-1.69e-4*Age$^2$ ), 1.24 )<br><br>Female:<br>LN( exp( 3.94+1.51e-2*Age-1.51e-4*Age$^2$ ), 1.28 ) | Age, Gender | EPA, 2009a | 8-4, p. 8-12;, 8-5, p. 8-13 | |
| Ventilation rate: sleeping | m3/min-kg | Male, age 16-20:<br>LN( 6.91e-5, 1.24 )<br><br>Male, age 21-60:<br>LN( exp( -9.91+4.93e-3*Age ), 1.26 )<br><br>Female, age 16-20:<br>LN( 6.71e-5, 1.29 )<br><br>Female, age 21-60:<br>LN( exp( -9.93+3.57e-3*Age ), 1.30 ) | Age, Gender, units in terms of Body Weight | EPA, 2009a, EPA, 2009b | 6-13, p. 6-33;, 6-14, p. 6-35 | |
| Ventilation rate: sedentary activity | m3/min-kg | Male, age 16-20:<br>LN( 7.58e-5, 1.20 )<br><br>Male, age 21-60:<br>LN( exp( -9.82+5.14e-3*Age ), 1.19 )<br><br>Female, age 16-20:<br>LN( 7.37e-5, 1.23 )<br><br>Female, age 21-60:<br>LN( exp( -9.86+3.89e-3*Age ), 1.24 ) | Age, Gender, units in terms of Body Weight | EPA, 2009a, EPA, 2009b | 6-13, p. 6-33;, 6-14, p. 6-35 | |
| Ventilation rate: light activity | m3/min-kg | Male, age 16-20:<br>LN( 1.77e-4, 1.18 )<br><br>Male, age 21-60:<br>LN( exp( -8.82+2.01e-3*Age ), 1.17 )<br><br>Female, age 16-20:<br>LN( 1.72e-4, 1.18 )<br><br>Female, age 21-60:<br>LN( exp( -8.88+2.55e-3*Age ), 1.20 ) | Age, Gender, units in terms of Body Weight | EPA, 2009a, EPA, 2009b | 6-13, p. 6-33;, 6-14, p. 6-35 | |
| Ventilation rate: moderate activity | m3/min-kg | Male, age 16-20:<br>LN( 3.80e-4, 1.21 )<br><br>Male, age 21-60:<br>LN( exp( -8.02+1.93e-3*Age ), 1.25 )<br><br>Female, age 16-20:<br>LN( 3.56e-4, 1.21 )<br><br>Female, age 21-60:<br>LN( exp( -8.10+1.40e-3*Age ), 1.25 ) | Age, Gender, units in terms of Body Weight | EPA, 2009a, EPA, 2009b | 6-13, p. 6-34; 6-14, p. 6-36 | |
| Ventilation rate: high activity | m3/min-kg | Male, age 16-20:<br>LN( 6.92e-4, 1.25 )<br><br>Male, age 21-60:<br>LN( exp( -7.38+5.56e-4*Age | Age, Gender, units in terms of Body Weight | EPA, 2009a, EPA, 2009b | 6-13, p. 6-34; 6-14, p. 6-36 | |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| | | ), 1.27 )<br><br>Female, age 16-20:<br>LN( 6.76e-4, 1.27 )<br><br>Female, age 21-60:<br>LN( exp( -7.37-4.88e-4*Age ), 1.30 ) | | | | |
| Adult incidental soil ingestion rate | mg/d | Silicon:<br>LN( 12.2, 3.29 ),<br>truncated at 0 and 197<br><br>Aluminum:<br>LN( 32.7, 3.81 ),<br>truncated 0 and 814<br><br>Titanium:<br>LN( 296, 2.76 ),<br>truncated at 0 and 2900 | Selection of tracer element performed outside of the "inner loop" | EPA, 2009a; Davis et al, 2006. | 5-11, p. 5-37 | Only study with applicable adult data. Truncation maxima based upon maxima reported in Davis et al, 2006, as pathological soil ingestion is not of interest here. |
| Ingestion rate: "home-produced" beef | g/kg-d | Age 16-39:<br>Gamma( 2.12, 1.77 )<br><br>Age 40-60:<br>Gamma( 1.89, 1.39 ) | Age, units in terms of Body Weight | EPA, 2009a | 13-33, p. 13-40 | |
| Ingestion rate: "home-produced" game | g/kg-d | Age 16-39:<br>Gamma( 0.84, 0.68 )<br><br>Age 40-60:<br>Gamma( 0.99, 0.83 ) | Age, units in terms of Body Weight | EPA, 2009a | 13-41, p. 13-48 | |
| Daily exposure time; sedentary+sleeping | hr/day | Males:<br>LN( exp( 2.79-1.55e-2*Age+2.09e-4*$Age^2$ ), 1.09 )<br><br>Females:<br>LN( exp( 2.84-1.71e-2*Age+2.10e-4*$Age^2$ ), 1.08 )<br>Truncated at 24 hr/day | Age, Gender | EPA, 2009a, EPA, 2009b | 6-15, p. 6-37 | Sedentary duration alone constructed by subtracting sleeping time. |
| Daily exposure time; sleeping | hr/day | Males:<br>LN( exp( 2.31-1.01e-2*Age+1.05e-4*$Age^2$ ), 1.06 )<br><br>Females:<br>LN( exp( 2.35-9.94e-3*Age+9.94e-5*$Age^2$ ), 1.06 )<br>Truncated at Sedentary+Sleeping time | Age, Gender, Sedentary+Sleeping time | EPA, 2009a, EPA, 2009b | 6-15, p. 6-37 | Sleep duration is excluded for daily-use receptors. |
| Daily exposure time; light activity | hr/day (un-normalized) | Males:<br>LN( exp( 2.38-3.44e-2*Age+4.05e-4*$Age^2$ ), 1.49 )<br><br>Females:<br>LN( exp( 2.09-1.37e-2*Age+1.69e-4*$Age^2$ ), 1.34 ) | Age, Gender | EPA, 2009a, EPA, 2009b | 6-15, p. 6-37 | Light, moderate, and high activities are normalized to equal: 24 hr/day – (sedentary + sleeping time). |
| Daily exposure time; moderate activity | hr/day (un-normalized) | Males:<br>LN( exp( 1.86e-1+6.74e-2*Age-8.16e-4*$Age^2$ ), 1.88 ) | Age, Gender | EPA, 2009a, EPA, 2009b | 6-15, p. 6-38 | Light, moderate, and high activities are normalized to equal: 24 hr/day – |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| | | Females:<br>LN( exp( 2.21e-1+6.49e-2*Age-7.85e-4*Age$^2$ ), 1.65 ) | | | | (sedentary + seeping time). |
| Daily exposure time; high activity | hr/day (un-normalized) | Males:<br>LN( exp( -1.12-2.19e-2*Age+3.14e-4*Age$^2$ ), 3.04 )<br><br>Females:<br>LN( exp( -1.97+4.04e-3*Age+6.27e-5*Age$^2$ ), 2.84 ) | Age, Gender | EPA, 2009a, EPA, 2009b | 6-15, p. 6-38 | Light, moderate, and high activities are normalized to equal: 24 hr/day – (sedentary + sleeping time). |
| Total number of individuals in vicinity of site | # | TRI(100, 350, 500) | | BLM, personal communication, 2010 | | Assumes area up to approximately 100 sq mi around site. This value, minus the number of ranchers (see text), defines the number of Sport OHVers and Hunters |
| Number of Ranchers in vicinity of site | # | U(1, 20) | | BLM, personal communication, 2010 | | |
| Number of Hunters in vicinity of site | # | Binomial( N, 0.25 ), where N is the number of non-rancher individuals in vicinity of site | Total number of individuals, number of ranchers | USFS, 2005 | 22, p. 32 | "Big game" hunters, all OHV users. Rounded to two significant figures. |
| Number of Sport OHVers in vicinity of site | # | Number(Recreationalists) - Number(Hunter) | Total number of individuals, number of ranchers and hunters | | | Number of Recreationists defined as all individuals minus Ranchers. |
| Ranchers; day trip time in exposure area | hr/d | U(4, 12) | | | | Professional judgment. |
| Sport OHVers; day trip time in exposure area | hr/d | Beta(6.3, 2.11, 1, 20) | | Burr et al, 2008 | 21, p. 18 | Utah data. Minimum, maximum, and standard deviation based upon professional judgment. Rounded to two significant figures. |
| Hunter/Rancher; fraction of day trip time spent OHVing | fraction | U(0.1, 0.75) | | | | Professional judgment. OHV use related to higher dust concentrations in air. |
| All receptors; camp trip time spent OHVing | hr/d | U(2.0, 8.0) | | | | Professional judgment. All overnight users assumed to have similar OHV use. OHV use related to higher dust concentrations in air. |
| Exposure time; overnight trip | hr/d | 24 | | | | Professional judgment; overnight trip assigned a 24 hr duration. |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| All receptors; fraction of camp trip exposure time on disposal cell | fraction | U(0.25, 0.75) | | | | Professional judgment. Corresponds to 6 to 18 hr/day. Campers are assumed to set up camp on the disposal cell. |
| Hunter; fraction of hunting day trip exposure time on disposal cell | fraction | U(0.02, 0.17) | | | | Professional judgment. Corresponds to 0.5 to 4 hr/day. |
| Rancher and Sport OHVer; fraction of day trip exposure time on disposal cell | fraction | Disposal cell area / Exposure area | | | | Assumes that Ranchers and Sport OHVers visiting the area for a day trip cover the exposure area randomly over the course of a year. |
| Rancher; exposure frequency | d/yr | Beta( 135, 34.9, 0, 180 ) | | BLM, personal communication , 2010; BLM, 2010 | | All leases are 6 mo., from November 1 to April 30, but can be reduced depending upon grazing conditions. It is assumed that Ranchers only work 5 days per week (i.e. 130 days per year). distribution based upon professional judgment. |
| Sport OHVer; exposure frequency | d/yr | LN(11.3, 3.45, 1, 200) | | USFS, 2005 | 19, p. 27 | Western region, "all groups". Minimum and and maximum based upon professional judgment. |
| Hunter; exposure frequency | d/yr | LN(4.66, 3.45, 1, 100) | | USFWS, 2006 | pg. 10 | Utah data. Recreationists who are not Hunters are defined as Sport OHVers: # Sport OHVers = # Recreationists in total - # Hunters. Mean calculated based upon number of hunters and days of hunting. Minimum, maximum, and standard deviation based upon professional judgment. |
| Ranchers; fraction of exposure frequency related to overnight trips | fraction | U(0.5, 0.67) | | BLM, personal communication , 2010 | | Corresponds to 15 – 20 day/month overnight. Remaining days in ranching EF assumed to be day trips. |
| Hunters; fraction of exposure frequency | fraction | U(0, 1.0) | | | | Professional judgment. |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| related to overnight trips | | | | | | |
| Sport OHVers; fraction of exposure frequency related to overnight trips | fraction | U(0, 1.0) | | | | Professional judgment. |
| **Off-Site Receptor Distributions (“Inner Loop”)** | | | | | | |
| Exposure frequency rest area caretaker | d/yr | TRI(327,350,365) | | | | Professional judgment. Minimum represents 28 days of vacation, 10 holidays, mode is EPA default (EPA, 1989), high is maximum. |
| Exposure time rest area caretaker | hrs/day | 24 | | | | Professional judgment (residential receptor). |
| Exposure frequency I-80 and west-side access road traveller | d/yr | U(250, 365) | | | | Professional judgment (minimum reflects average number of work days per year). |
| Exposure time travelers on I-80 and train | min/d | U(2.3, 7.2) | | | | Professional judgment. Minimum represents 80 mph/3 miles 1-way; maximum 50 mph/3 miles 2-way. 3 miles represents 'densest' part of off-site dispersion plume. |
| Exposure time cars on west-side access road (Utah Test and Training Range access) | min/d | U(2.4,4.0) | | | | Professional judgment. Minimum represents 50 mph/1 mile 2-way upper 30 mph/1 mile 2-way. 1 mile represents size of ES property. |
| Knolls area Sport OHVer; exposure frequency | d/yr | LN(11.3, 3.45, 1, 200) | | USFS, 2005 | 19, p. 27 | Western region, "all groups". Minimum and maximum based upon professional judgment. |
| Knolls area Sport OHVers; exposure time | hr/d | Beta(6.3, 2.11, 1, 20) | | Burr et al, 2008 | 21, p.18 | Utah data. Minimum, maximum, and standard deviation based upon professional judgment. Rounded to two significant figures. |
| **“Outer Loop” human exposure factors; sampled once each model realization** | | | | | | |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| Receptor area (exposure area) | acres | U(16000,64000) | | BLM, personal communication, 2010; BLM, 2010 | | Professional judgment. High-end reflects area between I-80 and UTTR, bounded by salt flats and Cedar Mt foothills. Low-end reflects Aragonite and E. Grassy range leases. This defines the exposure area for ranching and recreational receptors. |
| Meat preparation loss | fraction | N(0.27,0.07, 0.01, 1) | | EPA, 1997b | 13-5 | Converted from fractions. Fraction of meat (which is based upon beef, uncooked weight) lost in preparation. Minimum and maximum based upon professional judgment. |
| Meat post-cooking loss | fraction | N(0.24, 0.09, 0.01, 1) | | EPA, 1997b | 13-5 | Converted from fractions. Fraction of meat (which is based upon beef, uncooked weight) lost in preparation. Minimum and maximum based upon professional judgment. |
| OHV dust loading | multiplier for ambient dust concentration | LN(98.1, 1.65) | | EPA, 2008 | 2 | Activity based; i.e. OHVs generate increased dust. |
| Exposure frequency; food | d/yr | 365 | | EPA, 1997b | | Food intake rates are annual averages. |
| Soil ingestion tracer element | | Discrete(0.333) | | | | Professional judgment; equal probability assigned to distributions based upon aluminum, silicon, and titanium. |
| **Cattle and game radionuclide uptake exposure factors ("Outer Loop")** | | | | | | |
| Cattle range area, per operation | acres | | | | | See 'outer loop' parameter definition for Receptor area (exposure area). |
| Pronghorn range area | acres | U(995, 9192) | | Huffman, 2004 | | Foraging distances for summer and winter were equally weighted and assigned as diameters of a circular home range, from 0.1-0.8 km in the spring and summer to 3.2-9.7 km in the fall and winter. |

| Parameter | Units | Value | Dependencies | Source | Table | Notes |
|---|---|---|---|---|---|---|
| Cattle beef transfer factor | Bq/kg per Bq/d | (element-specific; see Table 3) | | IAEA, 2010; and others | | Also applied to pronghorn. |
| Cattle water ingestion rate | kg/day | U(33, 53) | | MSUE, 2011 | | Range of average daily water intake for "finishing cattle" of weights 600 – 1200 lb is 8.6 to 14 gallons. |
| Cattle forage ingestion rate | kg/day | U(8.85, 14.75) | | EPA 2005 | B-3-10, p. B-138 | Recommended value is 11.8 kg/day; range of +/- 25% is professional judgment. Value is dry weight. |
| Cattle soil ingestion rate | kg/day | U(0.05, 0.95) | | EPA 2005 | B-3-10, p. B-139 | Recommended value is 0.5 kg/day; range of +/- 100% is professional judgment. |
| Cattle time fraction in exposure area | fraction | Discrete(1.0) | | | | Professional judgment. Time grazing around the site is presumed to be sufficient to reach the equilibrium represented by transfer factors. |
| Pronghorn water ingestion rate | kg/day | U(0.1, 1) | | UDWR, 2009 | p. 4 | Professional judgment. Pronghorn may drink no water at all when fresh browse is available and up to 0.79 gal/day (3.0 L) during dry periods. Maximum set at 1 L/day. |
| Pronghorn body weight | kg | U(38, 41) | | Huffman, 2004 | | |
| Pronghorn forage ingestion rate | kg/day | 0.577 x Body Weight Factor$^{0.727}$ x 0.001 | | EPA, 1993b | Equation 3-9, p. 3-6 | Allometric scaling based upon body weight for mammalian herbivore. Units converted to kg/d. |
| Pronghorn soil ingestion rate | kg/day | U(0.005, 0.095) | | | | Professional judgment. Set equal to 10% of soil ingestion distribution for cattle based upon body mass. |
| **Plant ingestion screening calculations exposure factors ("Outer Loop")** | | | | | | |
| Dry-wet plant weight conversion factor | fraction | U (0.05, 0.30) | | EPA, 2009a | 9-33, p. 9-59 | Professional judgment. Based upon approximate range of moisture contents for edible parts of fruits and vegetables. |

# 2.0 Purpose and Context

A radioactive waste disposal facility located in Clive, Utah (the “Clive facility”) and operated by Energy*Solutions* LLC is proposed to receive and store depleted uranium (DU) and associated contaminants (called "DU waste" here). To assess whether the proposed Clive facility location and containment technologies are suitable for protection of human health, specific performance objectives for land disposal of radioactive waste set forth in Utah Administrative Code (UAC) Rule R313-25-8 (Utah, 2010) must be met. In order to support the required radiological PA, a detailed computer model has been developed to evaluate the potential future radiation doses to human receptors that may result from the disposal of DU waste, and conversely to determine how much DU waste can be safely disposed at the Clive facility.

The site conditions, chemical and radiological characteristics of the wastes, contaminant transport pathways, and potential human receptors and exposure routes at the Clive facility that are used to structure the quantitative PA model are described in the conceptual site model (CSM) documented in the *Conceptual Site Model for Disposal of Depleted Uranium at the Clive Facility* white paper (Appendix 2). The PA model has been developed as a probabilistic model taking into account site-specific conditions and uncertainties inherent to model variables (termed "parameters" here). The GoldSim systems analysis software (GTG, 2010) was used to construct the probabilistic PA model. This software supports probabilistic analysis of the release and transport of radionuclides from disposal systems. The PA model is intended to reflect the current state of knowledge with respect to the proposed DU disposal, and to support environmental decision making in light of inherent uncertainties.

The dynamic aspects of the PA model may be grouped into two domains. The 'contaminant transport' (CT) component of the PA model encompasses the release of contaminants from disposed wastes and subsequent migration through the environment. The output of the CT component (documented in other white papers) is a time series of contaminant concentrations in different environmental media. These concentrations serve as inputs to the 'exposure-dose' (ED) component of the PA model that is the subject of this white paper. Because the ED component of the PA model is organized within a single "container" in GoldSim, the terms ED model and ED container are used interchangeably.

Assumptions and mathematical equations describing contaminant intake, including external exposure to ionizing radiation, for each exposure scenario are provided here. Equations for estimating radionuclide dose, and non-carcinogenic toxicity associated with uranium, are also provided. The implementation of methods for evaluating uncertainty in the ED calculations are also described. The bases of the deterministic values and/or statistical distributions for each of the ED parameters are discussed in the text below, the attached Appendix I, the spreadsheet *Dose Assessment Appendix II*, and the *Model Parameters* white paper (Appendix 16).

# 3.0 Exposure-Dose Model Implementation

## 3.1 Summary of Exposure-Dose Model Scope

The ED container addresses potential radiation exposure, dose and non-carcinogenic toxicity to human receptors who may come in contact with contaminants released from the disposal facility into the environment subsequent to facility closure. Radiation dose limits for protection of the

general population are defined in UAC Rule R313-25-8 (Utah, 2010), and in 10 CFR 61.41 (CFR, 2007). These dose limits implicitly assume a level of health risk (discussed further below). The regulations specify that design, operation, and closure of the land disposal facility must also ensure protection of individuals inadvertently intruding into the disposal site and occupying the site or contacting the waste at any time after loss of active institutional control (e.g., fences, guards, etc.) of the site. Because the definition of inadvertent human intruders (IHI) encompasses exposure of individuals who engage in normal activities without knowing that they are receiving radiation exposure, there is no practical distinction made between a member of the public (MOP) and IHI with regard to receptors and dose calculations.

The UAC Rule R313-25-8 (Utah, 2010) requires a PA for DU to have a minimum compliance period of 10,000 years, with additional simulations for a "qualitative analysis" (i.e., one in which only contaminant migration, and not doses, are modeled) for the period where peak hypothetical dose occurs. The estimation of doses in such long time horizons would be speculative at best, but if total radioactivity is used for a proxy (accounting for radiological decay and ingrowth from the disposed DU), then a peak value would occur once the progeny of U-238 have reached secular equilibrium in about 2.5 million years. With respect to radiation dose and non-carcinogenic uranium toxicity, the ED container quantifies dose only within the regulatory time frame of 10,000 yr. This approach is consistent with the requirements of UAC R313-25-8 (Utah, 2010). No specific time frame is defined in 10 CFR 61 (CFR, 2007) for the exposure/dose assessment.

Key land use characteristics of the Clive facility that pertain to the development of receptor scenarios and dose modeling are summarized in the CSM (Appendix 2) and in the *Features, Events, and Processes (FEPs) Analysis for Disposal of Depleted Uranium at the Clive Facility* white paper (Appendix 1). Current human use of the area surrounding the Clive facility is very limited. Note that a residential scenario is not evaluated here, as there is no evidence that humans have permanently resided at the immediate Clive facility environs in recent history (see CSM). The closest current dwelling is approximately 12 km to the northeast of the site (a caretaker at the Aragonite/Grassy Mountain rest stop on east-bound Interstate-80).

Rancher and recreationist scenarios for the area surrounding the Clive facility are conditioned only on a continuation of present-day land use, whereas the conditions related to other scenarios would be much more speculative. It is not possible to project changes in human biology, society, technology, or behavior over a 10,000 year time frame; thus, current land use characteristics are projected throughout this period of performance, as recommended in NRC (2000). Uncertainty associated with this assumption is not quantified at this time. However, general justifications for this assumption in addition to NRC guidance can be made. The Clive facility environs are currently not amenable to permanent habitation due to the lack of potable groundwater and other factors. Dramatic changes in climate, such as large increases in average annual temperature or decreases in precipitation, would make the site even less hospitable. Changes in the opposite direction; i.e., large decreases in average annual temperature or increases in precipitation, have historically only been associated with ice ages and thus again would result in the site becoming less hospitable than it is today (see the CSM). Therefore, the assumption that future land use and receptors will be similar to today's is likely conservative (i.e., protective).

It is possible that the Clive facility disposal cap could become more amenable to plant cover and perhaps increased human use than the surrounding areas post-closure due to the presence of the rip rap cover (e.g., in terms of accumulation of aeolian or wind-borne soil and dust and lower evaporation rates from soil below the rip rap). Nearby areas hosting vegetation (e.g., the alluvial

fan of the Cedar Mountains east of the Clive facility, rocky outcrops west of the site) thus potentially offer analogous sites that will be considered for characterizing potential future plant communities on the disposal cap.

## 3.2 Exposure Scenarios

Based upon current and reasonably anticipated future land uses as summarized above, and as described in the FEP analysis (Appendix 1), two future use exposure scenarios were identified for inclusion in the ED model: ranching and recreation. After institutional controls are no longer maintained, exposures to contamination in the ranching and recreation scenarios could occur both on the Clive facility site as well as nearby off-site locations.

Modeling of ranching and recreation scenarios is discussed here. Exposure scenarios are defined according to various human activities, which may result in a complete exposure pathway existing between the contaminant source and receptors. Exposure pathways describe the media, activities and exposure routes by which contamination becomes available to human receptors in the exposure scenarios. Every complete exposure pathway contains the following elements (EPA, 1989):

- Known or potential sources and/or releases of contamination;
- Contaminant transport pathways;
- Potential exposure media;
- A point of potential receptor contact with the impacted medium; and,
- An exposure route (such as ingestion or inhalation).

The primary exposure routes for the ranching and recreation scenarios include ingestion, inhalation, and external irradiation. A summary of potentially complete exposure pathways for each scenario is provided in Table 2. Figure 10 in the CSM (Appendix 2) depicts the transport mechanisms by which contaminants in the disposed waste may reach the exposure media discussed in this section.

**Table 2. Exposure pathways summary**

| *Exposure Pathway* | *Ranching* | *Recreation* |
|---|---|---|
| Inhalation (wind derived dust) | × | × |
| Inhalation (mechanically-generated dust) | × | × |
| Inhalation (gas phase radionuclides) | × | × |
| Ingestion of surface soils (inadvertent) | × | × |
| Ingestion of game meat | | × |
| Ingestion of beef | × | |
| Ingestion of wild plant material | ×* | ×* |
| Ingestion of seasonal surface water | ×* | ×* |

| Exposure Pathway | Ranching | Recreation |
|---|---|---|
| External irradiation – soil | × | × |
| External irradiation – immersion in air | × | × |

*Not included in the ranching or recreation scenarios; see text.

Note that a single individual could potentially engage in both ranching and recreation in the same area, but these scenarios are modeled separately because they are expected to be distinct. Groundwater ingestion is not directly evaluated in the ED model, although groundwater concentrations are compared to State of Utah Ground Water Protection Levels (GWPLs). As described in the CSM (Appendix 2), the aquifers underlying the area are more saline than seawater, and would not be potable without extensive desalinization. This situation is unlikely to change under any foreseeable conditions that would allow human habitation in the vicinity of the facility.

It is possible that humans may be exposed by ingestion of native plants. Several plants identified in Clive area vegetation plots were historically used as traditional food or medicine. These include shadscale saltbrush (*Atriplex confertifolia*), black greasewood (*Sarcobatus vermiculatus*), and rockcress (*Arabis* sp.), among others. However, present-day use of these plants by potential receptors in the area is unknown. In the absence of such information for plant uses and quantities thereof, a screening-level calculation will be performed to determine what quantity of plant material from the disposal cap would need to be consumed to exceed the radiation dose performance objective.

A second possible exposure pathway not directly assessed in the ranching and recreation scenarios is human ingestion of intermittent (seasonal) surface water from puddles that may form in the air dispersion area. This surface water is likely to be salty, due to the saline nature of soils adjacent to the Clive facility, and direct human exposure is considered to be unlikely. Although present-day use of surface water by potential receptors in the area is unknown, a screening-level calculation will be performed to determine what volume of water would need to be consumed to exceed the radiation dose performance objective.

### 3.2.1 Ranching

The land surrounding the Clive facility is currently utilized for cattle and sheep grazing (BLM, 2010). Livestock apparently utilize the area more during winter periods when snow is present and when puddles exist during wet periods (NRC, 1993). The Bureau of Land Management (BLM) currently issues leases for 6 months of the year (November 1 to April 30; BLM, 2010, personal communication: Salt Lake Field Office). The personnel who spend time with the herds in the field are called "Ranchers" here (although this may include a variety of job classifications). Activities are expected to include herding, maintenance of fencing and other infrastructure, and assistance in calving and weaning. Ranchers may be exposed to contamination via the routes outlined in Table 1. It is assumed that any future ranching-related structures that might be constructed will be rough-built, with sufficient air flow that indoor radon accumulation is not an issue.

Ranchers typically use off-highway vehicles (OHVs; including four-wheel drive trucks) for transport. Beef consumption (from cattle exposed to contamination released from the site), is evaluated for the Ranchers, assuming that they may consume some of their own product. Beef,

rather than lamb or mutton, is used as a food in the ED ranching scenario because regulatory bodies such as EPA (2005) and others have published information related to modeling of tissue concentrations for cattle.

### 3.2.2 Recreation

The recreational exposure scenario could potentially encompass a variety of activities. Information is limited regarding current use, as the BLM, the manager of much of the surrounding land, does not specifically track recreational usage in the area. However, based upon discussions with the BLM and reasonable judgment regarding anticipated land use, recreation may involve OHV use, hunting, target shooting of inanimate objects, rock-hounding, wild-horse viewing, and limited camping.

The desirability of recreational activities on or around the disposal units, similar to suitability for ranching, is partially dependent upon assumptions regarding ecological succession on the disposal unit over time. With the possible exceptions of OHV use and as a vantage for hunting (e.g., for pronghorn), recreational use of the disposal unit in an as-closed state is likely to be minimal. As plant succession proceeds the disposal unit may become more attractive for different types of recreational activities. However, for the purpose of exposure assessment, it is assumed that sport OHV riders ("Sport OHVers; i.e., OHV users who use their vehicles for recreation alone) and hunters using OHVs ("Hunters"), both of whom may also camp at the site, would represent the most highly-exposed receptors (due to exposure to mechanically-generated dust, game meat ingestion, etc.), and other types of recreationists would have lower exposures.

### 3.2.3 Other Potential Receptors

The ranching and recreation scenarios are characterized by potential exposure related to activities both on the disposal site and in the adjoining area. Specific off-site points of potential exposure also exist for other receptors based upon present-day conditions and infrastructure. These locations and receptors include:

- Travelers on Interstate-80, which passes 4 km to the north of the site;
- Travelers on the main east-west rail line, which passes 2 km to the north of the site;
- Workers at the Utah Test and Training Range (a military facility) to the south of the Clive facility, who may occasionally drive on a gravel road immediately to the west of the Clive facility fenceline;
- The resident caretaker at the east-bound Interstate-80 rest facility (the Grassy Mountain Rest Area at Aragonite) approximately 12 km northeast of the site, and,
- Sport OHV enthusiasts at the Knolls OHV area (BLM land that is specifically managed for OHV recreation) 12 km to the west of the site.

Exposure to individuals at these off-site locations is expected to be minimal due to either the large distance from the site (Interstate-80 rest area and Knolls OHV area) or because the exposure time for any individual will be very brief (travelers on road, rail, and highway). Unlike ranching and recreational receptors who may be exposed by a variety of pathways on or adjacent to the site, these off-site receptors would likely only be exposed to wind-dispersed contamination, for which inhalation exposures are likely to predominate. These receptors will be evaluated to determine whether exposures at these off-site locations may be important.

## 3.3 Assessment Endpoints

The biological effect of greatest interest to regulatory agencies for environmental exposure to radionuclides is cancer. Ionizing radiation is a clear cause of cancer and other health effects at high doses. However, the risk of cancer to an individual exposed to radiation at environmental levels is highly uncertain and depends upon a large number of assumptions, the most influential being: 1) That the major source of data for radiological risk assessment; i.e., the high doses experienced by the Hiroshima/Nagasaki atomic bomb victims in World War II, is relevant for the much lower doses in the range of regulatory dose limits; and, 2) that risks can be extrapolated from large doses to small doses in a linear fashion, with no threshold of effect (i.e., the hypothesis that no dose is without some risk of cancer) (Brenner et al., 2003). Both of these assumptions are controversial (Scott, 2008), but they provide substantive bases for NRC and DOE radiation regulation and guidance at this time. Uncertainty associated with these assumptions is not evaluated in the PA model at this time.

### 3.3.1 Individual Dose

There are two performance goals that may be applicable in the PA. The first is the individual dose limit. Title 10 CFR 61.41 (CFR, 2007) specifies assessment endpoints for a radiological PA that are related to annual radiation dose. The specific metrics described in §61.41 are organ-specific doses, and restrict the annual dose to an equivalent of 0.25 mSv (25 mrem) to the whole body, 0.75 mSv (75 mrem) to the thyroid, and 0.25 mSv (25 mrem) to any other organ. As described below, the ED model will employ a total effective dose equivalent (TEDE) for comparison with the 0.25 mSv/yr threshold. This dose level will be considered as a deterministic performance goal, with no uncertainty.

As discussed in Section 3.3.7.1.2 of NUREG-1573 (NRC, 2000), the radiation dosimetry underlying the §61.41 dose metrics was based upon a methodology published by the International Commission on Radiation Protection (ICRP) in 1959. Subsequent to Title 10 CFR 61.41, more recent dose assessment methodology has been published by the ICRP (ICRP, 1979; 1991; 1995) that employs the TEDE approach. The TEDE uses weighting factors related to the radiosensitivity of each target organ to arrive at an effective dose equivalent across all organs. The text of Section 3.3.7.1.2 of NUREG-1573 (NRC, 2000) states:

> *As a matter of policy, the Commission considers 0.25 mSv/year (25 mrem/year) TEDE as the appropriate dose limit to compare with the range of potential doses represented by the older limits... Applicants do not need to consider organ doses individually because the low value of TEDE should ensure that no organ dose will exceed 0.50 mSv/year (50 mrem/year).*

The regulations state that this dose limit is applicable to *any* member of the public, yet NRC PA guidance (NRC, 2000) suggests a practical approach of applying the dose limit to an *average* member of a "critical group" (i.e., a group of public receptors who might be reasonably expected to live near or experience exposure to the facility site). The ED model has been developed to support estimates of both average individual dose and various percentiles of the distribution of the mean individual dose for Ranchers, Sport OHVers, and Hunters at any model year of a simulation.

Thus, in terms of PA performance objectives, the modeling question relates to estimating the probability that the total radiation dose attributable to future releases from the site to any or an average member of a critical group (defined here as a Rancher, Sport OHVer, or Hunter) will exceed 25 mrem TEDE in any particular year, during the performance period of the site. As institutional controls in place while the site is operating are designed to prevent public access, there will be no public exposure during this time period. The period of time of interest, therefore, in the ED portion of the PA model is from the time of loss of institutional control to 10,000 years post-closure, although physical transport processes are evaluated beginning at model year zero.

The US Environmental Protection Agency (EPA) has estimated that 15 mrem/year is equivalent to a 3-in-10,000 excess risk of cancer (EPA, 1997a), and has defined that level as:

> *...consistent with levels generally considered protective in other governmental actions, particularly regulations and guidance developed by EPA in other radiation control programs.*

A 1- in-1-million excess risk level is typically viewed as a *de minimus* level; i.e. one that is below a level of concern (CFR, 1994). If the estimated EPA risk equivalence for 15 mrem/year is extrapolated to 1- in-1-million, this results in a 0.05 mrem/year *de minimus* dose. This is potentially important both when evaluating the dose to *any* receptor and when collective dose is assessed (discussed below).

### 3.3.2 As Low As Reasonably Achievable (ALARA)

A second decision rule pertains to the ALARA concept. Ionizing radiation protection limits have been utilized since the 1920s (Hendee and Edwards, 1987). These limits have changed over time as more information regarding the negative biological effects of radiation has become available (especially after World War II). Concurrently, therapeutic and diagnostic (i.e., beneficial) uses of radiation have increased dramatically. Radiation in high doses kills cells, which can be harmful or beneficial to the receptor of the doses (e.g., in the latter case, targeted radiation is used to kill cancer cells). The effects of low doses of radiation are more uncertain. There is ample evidence that ionizing radiation can damage DNA and enhance cell proliferation in doses below those that kill cells, and thus can potentially cause cancer. However, it is uncertain at dose this becomes a concern.

For many years, there has been a presumption in radiation protection, based upon statistical analysis of animal and human data, that ionizing radiation has a linear dose-response curve at low doses and that there is no threshold of effect; i.e. any dose of radiation can result in an increased probability of cancer (this is termed the linear no-threshold, or LNT, hypothesis). This is not supported by all experimental and clinical observation (Scott, 2008) and multiple highly-efficient molecular and cellular defense and repair mechanisms for radiation damage exist. Regardless, this LNT hypothesis is the basis for most regulatory standards today, and indeed for the ALARA concept.

ALARA (or the older but similar concept "as low as practicable"; ALAP) essentially assumes no carcinogenic threshold of radiation carcinogenesis. If this assumption is taken at face value, ALARA seems to be a reasonable objective. If not, then a threshold of effect would be a more tractable and achievable objective. ALARA could perhaps be applied even in the case of a threshold or 'target' concentration; the threshold would simply be a limit on the amount of risk reduction that should be achieved by a particular management alternative. Proper evaluation of

uncertainty associated with the LNT hypothesis would be a large task in itself, but the influence of a LNT assumption can still in principle be evaluated using sensitivity analysis.

A different sort of threshold exists with regard to natural background levels of radiation. The doses that the public receives from all environmental sources (e.g., local geology, extraterrestrial, etc.) can be quite variable. For example, population *X* who live at high altitude in a location with geologically high levels of uranium may have a much higher level of annual exposure than population *Y* who live at sea level with low levels of uranium in soil (e.g., see http://www.epa.gov/radon/zonemap.html). If population sizes were equivalent, one could then consider that a larger incremental dose might be acceptable for population *Y* compared to population *X*.

Uranium and many other metals are also associated with non-radiological toxicity; e.g. kidney or liver damage. In such cases, toxicology has developed concepts such as the reference dose and benchmark dose, to account for the clear thresholds of effect that are associated with non-carcinogenic toxicity (Filipsson, 2003). Similar to the discussion above, in these cases the threshold can be viewed as a target, below which risks are not of substantial concern.

The modern ALARA concept, as germane to radiation protection on both individual and population levels, was described by the ICRP in 1977 (ICRP, 1977):

> *Most decisions about human activities are based on an implicit form of balancing of costs and benefits leading to the conclusion that the conduct of a chosen practice is 'worthwhile.' Less generally, it is also recognized that the conduct of the chosen practice should be adjusted to maximize the benefit to the individual or to society. In radiation protection, it is becoming possible to formalize these broad decision-making procedures.*

The ICRP (1977) basically recommended a system of radiation protection that included the following principles:

- No practice shall be adopted unless its introduction produces a positive net benefit;
- All exposures shall be kept as low as reasonably achievable, economic and social factors being taken into account; and,
- The dose equivalent to individuals shall not exceed the limits recommended for the appropriate circumstances by the Commission.
- These three components are identified by the ICRP by the abbreviated terms:
- The justification of the practice;
- The optimization of radiation protection; and,
- The limits of individual dose equivalent.

For present purposes, as regulatory agencies have adopted and applied clear dose limits for individuals, evaluation of ALARA here will be restricted to population doses, termed collective dose. This is appropriate in the context of design and siting of radioactive waste facilities; as it is likely, if any substantial future risks occur, that health concerns will be at a population level. Further, we assume that facility workers will be protected under existing health and safety regulations and guidance, and will not be evaluated here.

ICRP 101b (2006) describes updates to previous ICRP publications addressing ALARA. Section 3.3.3 discusses calculation of collective dose in the context of this publication.

### 3.3.3 Collective Dose

In order to estimate collective dose, a population needs to be assessed. If cumulative doses are to be estimated over some period of time, then the doses are added over that time period. The collective dose at the end of the performance period (10,000 years post-closure, in this case) is then the individual annual doses added up over a period of 10,000 years (minus the period of time when institutional controls are in place).

For a hypothetical example, say a total population of 50 people is potentially exposed to the site for every year during the performance period (note that all radioactive waste repositories that have been recently evaluated in the US are in fairly remote areas, so a large urban population would be inappropriate). Say institutional controls are in place for 100 years. Then, the cumulative population dose will be the sum of 50 individual doses in mrem/year, multiplied by 9,900 years. Say that every person in the population is exposed just below the individual dose limit (say, 24 mrem/year TEDE). Thus, the cumulative population dose will be 50×24×9900=11,880,000 mrem, or 11,880 person-rem. This number has no meaning by itself, as there is no standard or basis for declaring this is 'unacceptable' or not, or whether it is "reasonable" or "achievable" (according to ALARA). It is only useful in the context of comparing how one site or disposal option might perform compared to another. This is best determined in the context of a decision or economic analysis, which is discussed in the *Decision Analysis (ALARA)* white paper (Appendix 12).

In lieu of guidance that defines what an 'acceptable' population dose might be, a means must be applied so that all populations (e.g., the entire United States) are not assessed, as this would be burdensome and meaningless. For instance, it is known that a large population will indeed be exposed to the site if current conditions continue; i.e., the population of drivers on Interstate-80. However, as previously mentioned, each of these drivers would be exposed for very short periods of time. Furthermore, the exposure levels would be a small fraction of those experienced by the Ranching and Recreation receptors described in Section 3.2. In order to gauge the importance of quantifying dose for this population, and indeed any remote population that might be exposed for brief periods and/or to very low concentrations, a *de minimus* risk approach will be considered. As explained previously, according to the EPA a 0.05 mrem/year dose corresponds to approximately a 1-in-1-million excess cancer risk. Individual doses for receptors other than Ranchers, Sport OHVers, or Hunters will be evaluated relative to this individual dose threshold to determine whether doses to remote receptors should be considered when computing collective dose. Cumulative population dose will not include contributions from remote receptors if individual doses for these receptors are far below 0.05 mrem/year.

Note that NRC was required under Section 10 of the Low-Level Waste Policy Amendments Act of 1985 to “establish standards for determining when radionuclides in waste streams were in sufficiently low concentrations or quantities as to be below regulatory concern, thereby potentially exempting them from NRC Low-Level Waste regulation” (NRC, 2007; NUREG-1853, Section 3.5). The *de minimus* risk level discussed above is in no way related to establishing concentrations or quantities “below regulatory concern” in disposed waste. Rather, this level is employed to support a methodology for meaningful evaluation of collective radiation dose in relation to the ALARA assessment endpoint of the Performance Assessment.

## 3.4 Modeling Doses

### 3.4.1 Individual Doses

Studies of the health of existing populations (i.e., epidemiological studies) have struggled with how to infer individual risk from population statistics. For example, a study of cigarette smokers and lung cancer may show a clear statistical relationship between the exposure and disease, with a high degree of confidence; yet, for instance, it does not tell *me* what *my* additional risk of cancer will be if I smoke one cigarette. It is indeed impossible to directly estimate health risk for individuals for the majority of exogenous exposures (there are exceptions in the case of some genetic abnormalities; if the abnormality is known to exist in an individual, then the risk of disease in that individual associated with that abnormality is known with almost perfect confidence). Risk for individuals must generally be inferred from populations. In addition to various designs of epidemiological studies, insurance companies, for example, use life tables stratified on gender, age, disease history, etc. to estimate premiums.

In the present case, the issue is estimation of individual radiation doses. As mentioned above, risk is implicit in radiation dose, with many inherent assumptions. Additionally, the PA is projecting into the future, to individuals who do not exist yet. As information as to how humans may or may not change biologically in the space of a 10,000-year performance period does not exist, it is only reasonable to assume that humans will remain essentially the same.

One approach to estimating individual risk, based upon how the EPA has historically conducted exposure assessment (EPA, 1989), is to define a 'simulated' individual based upon their exposure characteristics. The simulated individual is therefore the product of a number of physiological and behavioral parameters. Historically, this has been done deterministically; i.e., single values are used for the exposure and physiological parameters, and a single simulated individual results. With more recent applications of probabilistic methods, this process has been expanded to address variance in the exposure parameter values.

For the Clive facility, following are some major sources of variance related to radiation dose that are directly germane to the ED model at any particular point during the assessment time horizon:

1. The number of receptors, if any, in the vicinity of the disposal site at any point in time;
2. The physiological characteristics of the receptors;
3. The nature and intensity of exposure by various potential exposure routes (ingestion, inhalation, external radiation) based upon behavioral characteristics of the receptors;
4. The concentrations of radionuclides in potential exposure media; and,
5. The annual radiation dose associated with the exposure.

Within some of these five categories there may be multiple exposure parameters employed in the modeling and hence numerous sources of variance. In particular, radionuclide concentrations in exposure media include all the variance from the contaminant transport modeling conducted in the PA that are propagated to the ED assessment.

As discussed above, the PA guidance (NRC, 2000) suggests that the annual dose to an "average member of a critical group" should be estimated. Specifically:

> *The average member of the critical group is that individual who is assumed to represent the most likely exposure situation, based on cautious but reasonable exposure assumptions and parameter values. It is generally not practicable, when analyzing future potential doses, to calculate individual doses for each member of a critical group and then re-calculate the average dose to these same members. In general, it is more meaningful to designate a single hypothetical individual, representative of that critical group, who has habits and characteristics equal to the mean value of the various parameter ranges that define the critical group. In this fashion, the dose to the "average member" of the critical group approximates the average dose obtained if each member of the critical group were separately modeled and the results averaged.*

Thus, the guidance appears to request definition of:

- A critical group;
- An average member of the critical group; and,
- The annual dose to this member.

The critical groups, in the case of the present PA, are defined as Ranchers, Sport OHVers, and Hunters. An "average member" of these groups is a theoretical or statistical construct, as such a person does not and never will exist. Thus, we can interpret the guidance as referring to the statistical average dose (i.e., arithmetic mean) of a population of individuals' doses. In order to estimate the average simulated individual's dose at a particular time step, doses to a population of simulated individuals need to be estimated (note that hardware and software capabilities have increased dramatically since the NRC's guidance, so it is indeed now possible to calculate doses at an individual level).

In the context of human health risk assessment, variance in parameter values is traditionally split into the categories of variability and uncertainty (EPA, 2001). The term *variability* refers to natural, irreducible variance in the range of values a parameter may take (say, body weights in a population), and *uncertainty* refers to incomplete, imprecise and/or inaccurate knowledge associated with parameter values (Bogen et al., 2009). These particular definitions are not universally accepted however, and in practice may have more or less utility as a basis for the methodology used to assess overall variance in model output.

Returning to the issue of doses to a population of simulated individuals, and to the five major sources of variance for these dose estimates, the first 3 sources of variance apply to population variability. In particular, in any year the physiological and behavioral characteristics of the exposed individuals govern the degree of variance related to sources #2 and #3. The variance related to parameters contributing to exposure concentrations and to radiation dose coefficients do not vary over time and do not vary for different hypothetical individuals. For example, models of carcinogenesis for low-dose radiation are highly uncertain, but this uncertainty does not appreciably differ among individuals nor does it vary from one model year to another. Similarly, we assume essentially static environmental conditions over the 10,000-year performance period for any given model realization; a soil-water distribution coefficient that applies at model year 2,000 also applies at model year 3,000.

There are multiple methods that may be employed to model two different types of variance, but a typical method is termed 2-dimensional (2D) or nested-loop Monte Carlo simulation (Bogen et al., 2009). In the ED model, the exposure parameters are grouped into long-term model uncertainty and population variability categories. The physiological and behavioral parameters

related to sources #2 and #3, as well as the number of individuals exposed in any year (source #1), are evaluated annually in the "inner loop" of the 2D Monte Carlo simulation. The remainder of the model parameters, including all aspects of the Contaminant Transport modeling and the radiation dose conversion factors (DCFs) are defined in the "outer loop" of the 2D Monte Carlo simulation. This categorization is further discussed below.

### 3.4.2 Collective Dose

As described above, an issue of ALARA interest is the collective dose over the performance period. To reiterate, this estimate is of little value in itself as there are no performance objectives for this endpoint; rather, it should ideally be viewed in the context of decision analysis.

Estimating population dose is simple. It is the sum of individual annual doses over the period of time from loss of institutional control to the 10,000 year mark. Contributions from off-site receptors who are anticipated to have very low annual dose rates will only be included in the collective dose sum if individual doses are approaching a 0.05 mrem/yr threshold (equivalent to approximately a 1-in-1-million excess cancer risk).

The calculation of collective dose is consistent with recommendations of the ICRP (2006). For example, the PA's methodology specifically addresses the following characteristics of the population (ICRP, 2006; Table 3.1):

- Gender
- Age
- Habits
- Characteristics of the exposure
- Distribution of exposures in time and space
- Number of individuals
- Minimum individual dose
- Maximum individual dose
- Mean individual dose
- Statistical deviations
- Collective dose associated with ranges of individual doses.

### 3.4.3 Dose Conversion Factors

For both individual doses and population doses, exposures or intakes are converted to TEDEs via DCFs, or dose equivalents per unit intake. DCFs have been published by EPA and ICRP. Section 3.3.7.3 of NUREG-1573 specifies DCFs published by EPA in Federal Guidance Reports (FGR) 11 (EPA, 1988) and 12 (EPA, 1993a). EPA subsequently made use of age-specific DCFs published in ICRP Publication 72 (ICRP, 1995) to estimate radionuclide cancer risk coefficients in FGR 13 (EPA, 1999). The DCFs published in EPA (1999) are used in the dose assessment and are available online (http://ordose.ornl.gov/downloads.html). The radionuclide-specific DCFs used in the dose assessment are also provided in the spreadsheet *Dose Assessment Appendix II*.

DCFs are derived using models and data that represent the physics and biology of the interaction of the human body with radiation or radioactive material. Briefly, internal DCFs (typically in units of Sv/Bq) are used to convert from an exposure or intake to an internal dose delivered to

target organs. DCFs are radionuclide, receptor-age, and exposure-route dependent (external, inhalation, or ingestion). In addition, separate inhalation dose coefficients are published for different lung absorption rate classes. For external exposure the dose coefficient depends upon whether the receptor is immersed in a plume of radioactive contaminants (such as air) or is standing on the surface of contaminated ground (surface water sources are not evaluated here).

A number of groups have investigated uncertainty in radiation dose that is delivered to internal target organs (i.e., effective dose, via use of DCFs). For example, the US National Committee on Radiation Protection and Measurements (NCRP) has published a general methodological guide for uncertainty analysis in dose and risk assessments (NCRP 1996), a guide for evaluating the reliability of the biokinetic and dosimetric models used to assess individual doses (NCRP 1998), and assessments of uncertainties associated with internal (NCRP 2009) and external (NCRP 2007) dosimetry. Additionally, the United Kingdom's Health Protection Agency's (HPA's) Centre for Radiation has conducted uncertainty analyses of internal and external dosimetry (Puncher and Harrison 2012, 2013).

Major sources of uncertainty associated with effective dose estimation include the following (Puncher and Harrison 2012):

- Biokinetic models and their parameter values that are used to predict the dynamic distribution of radioactivity within the body
- The geometric relationship of source and target tissues, their dimensions and masses. These influence the amount of energy deposited in tissues
- The relative effectiveness of different radiation types in causing cancer and differences between tissues in their sensitivity to radiation induced cancer

Estimation of disease dose-response and risk (i.e., risk assessment) and associated uncertainties involves 'translating' effective dose into estimation of additional disease (typically cancer) probability. The Biological Effects of Ionizing Radiation (BEIR) VII report (National Research Council 2006) contains extensive information on the state of knowledge regarding radiation dose-response, including a limited uncertainty analysis. Both NCRP (2012) and EPA (EPA 2007) have investigated some sources of uncertainty in risk assessment.

With regard to evaluating radiation risk, major sources of uncertainty include the following (NCRP 2012):

- Issues associated with epidemiological and animal study design and application, including low statistical power and precision
- Inadequate or simplistic modeling of radiation risk (especially at low doses), or assumption of one generic model (typically the the linear no-threshold hypothesis, or LNT, model)
- Extrapolation or generalization of risk estimates to different populations

As an example, EPA (2007) estimated uncertainties for radionuclides that have published risk coefficients in EPA's Federal Guidance Report (FGR) No. 13 (EPA 1999). They addressed the following sources of uncertainty:

- Biokinetic models describing the biological behavior of ingested or inhaled radionuclides

- Specific energies that relate emissions from source organs to energy deposition in target organs
- Risk model coefficients representing the risk of cancer per unit absorbed dose to sensitive tissues from radiation at high dose and high dose rates
- Tissue-specific dose and dose rate effectiveness factors (DDREF); and tissue-specific high-dose relative biological effectiveness (RBE)

Uncertainties associated with alternative dose-response statistical models (i.e., aside from the LNT model) were not addressed by EPA (2007). EPA (2007) employed a combination of modeling and expert opinion in the analysis, and concluded that "the assessed uncertainty in the radiation risk [as opposed to dose] model was found to be the main determinant of the uncertainty category for most risk coefficients, but conclusions concerning the relative contributions of risk and dose models to the total uncertainty in a risk coefficient may depend strongly on the method of assessing uncertainties in the risk model".

All groups that have attempted to analyze uncertainties associated with radiation effective dose and risk have acknowledged that this is a difficult undertaking, and there is no generic "one-size-fits-all" solution. Each type of radiation and target organ dose-response has unique characteristics. Therefore, the most straightforward way to evaluate uncertainties in dose and risk may be to employ the FGR 13 central values and 'uncertainty categories' published by EPA (1999, 2007). These are represented as a ratio of the 95th to the 5th quantiles. As an example, if an uncertainty factor is 100, then a risk coefficient could vary from the published FGR 13 value by a factor as great as 10 (the square root of 100). Most radionuclides fall within categories A or B.

Unlike any other sources reviewed, ratios are available for a large (>800) number of radionuclides. The exact ratio values (as opposed to the letter categories) are available for all radionuclides with risk coefficients in FGR 13 (EPA 1999). Assuming a distributional shape such as lognormal, distributions can then be developed.

If uncertainties associated with effective dose only are evaluated (which is the approach taken at this time in the DU PA), the scope of existing and published work is much more limited. In order to be useful for probabilistic modeling, the uncertainties associated with DCFs must be represented as statistical distributions. A search of the published literature indicates that uncertainty distributions for DCFs *per se* have only been developed in a few instances; largely focused on a few radionuclides (e.g., I-131, tritium) that have been to focus of worker protection assessments, legal cases, and related dose reconstruction scenarios (e.g., Hamby, 1999; Harvey et al., 2006). Puncher and Harrison (2012, 2013) evaluated uncertainties for 9 radionuclides via ingestion and inhalation. For the purpose of the PA, uncertainty distributions for a large number of DCFs would ideally be available. No such 'global' source was identified in the literature.

However, there has been published work that has focused on components of DCFs that are generalizable to different classes of radionuclides. The most relevant work that was identified is the work of Kocher et al. (2005), in the context of "probability of causation" in cases of worker exposure to radiation. This work has been incorporated into the National Institute for Occupational Safety and Health's "Interactive RadioEpidemiological Program" (IREP; http://www.cdc.gov/niosh/ocas/ocasirep.html), which is employed to determine the probability that a cancer was caused by workers' exposure to radiation during nuclear weapons production.

Similar work has been applied in the context of probabilistic dose reconstruction (Linkov et al., 2001.)

Kocher et al. (2005) estimate:

> *…so-called radiation effectiveness factors (REFs) [note: not to be confused with 'radon emanation factors'] that are intended to represent the biological effectiveness of different types of ionizing radiation for the purpose of estimating cancer risks and probability of causation of radiogenic cancers in identified individuals. An REF is a dimensionless factor used to modify an estimate of average absorbed dose from a given radiation type in an organ or tissue of concern in an identified individual to obtain a biologically significant dose on which the risk of induction of cancer in that organ or tissue is assumed to depend.*

Kocher et al. (2005) specify that they are ultimately interested in *risks*, not *doses*; but, the estimates of uncertainty associated with REFs are relevant to the current application. They state that their REFs are essentially analogous to radiation weighting factors ($w_R$). The $w_R$ is is an additive function of a dimensionless "quality factor" *Q*, that is dependent upon radiation type; and a dimensionless *N*, which is dependent upon the tissues irradiated, the time and volume relevant to irradiation, and biological characteristics of the receptor. Consistent and thorough documentation of these terms appear to be lacking in published reports. Regardless, in most cases, these terms have been superseded by another term; Relative Biological Effectiveness (RBE). The radiation dose unit employed in this PA, the sievert (Sv), can vary considerably based upon the RBE.

Kocher et al. (2005) state that their

> *…new term "radiation effectiveness factor" (REF) is used in this work to distinguish a quantity that represents biological effectiveness for purposes of estimating cancer risks and probability of causation in identified individuals from similar quantities, including relative biological effectiveness (RBE), which strictly applies only to results of specific radiobiological studies under controlled conditions.*

For the purpose of establishing initial uncertainty distributions for DCFs for incorporation into the PA, these philosophical and semantic issues will take a subservient position. We will therefore assume that for the carcinogenic effects of radiation, that the REF is equivalent to the RBE, which is in turn equivalent to $w_R$. This is not strictly the case, but the intent here is to estimate uncertainty in biologically-relevant radiation dose, not exact numerical quantities. REFs account for the fact that some types of radioactive decay result in more biological damage than others. The "reference" type of radiation is typically Co-60 high-dose/dose-rate gamma decay, as this is the type of radiation germane to the atomic-bomb survivor data and similar sources of epidemiological data on cancer resulting from radiation exposure. The REF (or $w_R$) for such radiation is set at 1.0. However, larger particles such as alpha particles and neutrons can cause more biological damage, thus the REFs for these types of ionizing radiation are larger, and function as multipliers to the DCFs.

In this PA model, radiation-type specific REFs per Kocher et al. (2005) will be used as modifying distributions to the DCF point estimates presented in FGR 13 (note that DCFs are not presented in the written report of FGR 13, but are available via an online database: http://ordose.ornl.gov/downloads.html). Kocher et al. (2005) developed probability distributions

for REFs, based upon a combination of exhaustive literature review, statistical analysis, modeling, and subjective judgment. Tables 14 and 15 in that reference provide summaries.

These REF distributions can be essentially viewed as modifiers to published DCFs, in lieu of the published deterministic $w_R$'s used in radiation protection (ICRP, 1991). For example, the published deterministic $w_R$ for alpha particles is 20. The Kocher et al. (2005) REF for alpha particles can be represented by a lognormal distribution with a median of 18, and a 95% confidence interval from 3.4 to 100. Thus, for an alpha-emitting radionuclide, the published DCF would be divided by 20, then multiplied by the distribution provided. As the REFs are radiation-type specific, they are generally applicable to the predominant radiation characteristics of the particular radionuclide of concern.

In the present model, there are no species that decay by neutron emission. The REFs employed represent alpha, beta (electron), and photon (gamma, X-ray) decay. For each radionuclide, the dominant radiation type and its energy are defined based upon information from ICRP (using the program RadSum32, available from http://ordose.ornl.gov/downloads.html). For some radionuclides, the energy of electron or photon emissions is essentially equivalent to the reference radiation (high-energy gamma), resulting in an REF of 1.0 with no uncertainty. For others, an REF distribution is defined based upon the information in Kocher et al. (2005) and this REF is used as a multiplier to the DCF. Please note that radon is evaluated differently from other radionuclides (see Section 4.4); thus the REF distribution development process outlined below does not apply.

Following is a summary of the specific process by which REF distributions are generated and applied in the PA model, along with assumptions (please see Kocher et al. (2005) for assumptions made in that work). Radionuclide-specific deterministic DCFs, and the inputs necessary to calculate stochastic DCFs, are provided in the spreadsheet *Dose Assessment Appendix II*.

1. The 27 radionuclide Species in the PA model were expanded to 63 radionuclides to account for short-lived progeny (Species radionuclides have a half-life of approximately 2 years or longer). The decay chains for identifying progeny were taken from the Nuclear Wallet Cards (Tuli, 2005).

2. DCFs were taken from the the EPA FGR 13 database (available from http://ordose.ornl.gov/downloads.html). DCFs are available for particulate and vapor-phase inhalation, ingestion, and external exposure (including "submersion", "ground plane", and "soil volume" values). In all cases, DCFs for adults are selected (as the receptors of interest are adults), and "effective dose" DCFs (a weighted composite of all organs) are employed. Inhalation DCFs related to the default inhalation absorption class from Table 2.1 of FGR 13 were used. If no default class was specified, the "medium" (Class M) inhalation DCF was usually selected because it is commonly between the DCF values for slow and fast absorption classes, and is therefore considered to be the least biased point estimate. For external exposure to contaminated soils, the "soil volume" external DCFs are used in this PA consistent with the physical models of contaminant transport over time.

3. A dominant form of radiological decay was assigned for internal DCFs and external DCFs for each of the 63 radionuclides using information from the RadSum32 code. For internal DCFs, the dominant decay mode was identified as the highest contributor to total emitted energy of any radiation type (gamma + x-ray; electron (the maximum of beta, internal conversion electrons, or

auger electrons); and, alpha). In all cases, this protocol resulted in alpha emissions being selected as the dominant decay mode when alpha decay occurs. For external DCFs, the dominant decay mode was identified as the energy of gamma + x-ray. If there are no photon emissions for a radionuclide, dominant decay for external irradiation was identified as the highest energy among beta, internal conversion electrons, and auger electrons. Because alpha particles cannot penetrate the stratum corneum to the biologically active lower strata of the skin, alpha particles are not evaluated for the purpose of assigning REF distributions to external DCFs.

4. For radionuclides where the dominant decay mode is electron or photon, the average particle energy of that decay mode (in million electron volts, or MeV) is identified from the RadSum32 code.

5. REF distributions are defined for four categories of decay mode and energy, based upon percentiles in Tables 14 and 15 in Kocher et al. (2005). For radionuclides where the dominant decay mode is photon or electron emission with a mean energy higher than the particular threshold, an REF of 1.0 is assigned, as the REF for these emissions are essentially equivalent to the reference radiation (Co-60 gamma). The REF distribution categories include:

- alpha (any energy)
- electron (<0.015 MeV)
- photon (>0.03 and <=0.25 MeV)
- photon (<=0.03 MeV)

6. With regard to the alpha REF, please note that Kocher et al. (2005) assumed that 100 represented the 97.5th percentile of the distribution. This is likely conservative, as the highest value ever estimated from experimental studies is 100, and this only applies to particular forms of inhaled plutonium (Kocher et al., 2005).

7. The DCFs for each of the 63 radionuclides are divided by the ICRP weighting factor ($w_R$) in order to apply the REF distributions. For alpha emitters, the $w_R$ value is 20, and for electrons and photons it is 1.0. Stochastic DCFs are then calculated as the product of the DCF and the appropriate REF.

8. DCFs for the 27 radionuclide Species defined in the PA model are assembled using the decay chains and branching fractions from the Nuclear Wallet Cards (Tuli, 2005). These are equivalent to the "plus daughters" (+D) DCFs for primary radionuclides provided in radiological dose software such as the RESRAD computer code (http://web.ead.anl.gov/resrad/home2/).

9. The stochastic +D DCFs may then be employed in the PA model for radiation dose calculations. Alternatively, a model user may select the option of using the deterministic FGR 13 DCFs in a simulation. This is permitted even when the PA model is run in stochastic mode for all other model parameters.

As previously discussed, this method only addresses one component of uncertainty associated with DCFs, and thus must be viewed as a pilot effort. DCF distributions are available for some radionuclides, and could be incorporated into future modeling. Use of EPA (2007) risk coeffi-cients in addition to or in lieu of dose estimations would be a logical next step in expanding the scope of the uncertainty analysis for the health effects of radionuclides.

### 3.4.4 Additional Sources of Uncertainty

In addition to variance in the definition of model parameter values, there are other important sources of uncertainty and/or bias to potentially consider. For example, if radiation dose-response model uncertainty (particularly at low doses) were to be considered, it is possible that the uncertainties associated with radiation risk would swamp those associated with the remainder of the PA model, as it is by no means clear that ionizing radiation has *no* threshold of carcinogenic effect.

here is uncertainty associated with the mathematical models defining contaminant transport in the environment over time. These models are designed to represent the system as best they can (although sometimes with known protective biases) but they like all models are simply approximations of reality. Other aspects of the PA model have similar issues associated with model uncertainty.

Most importantly, the overall uncertainty associated with what the natural world and human society will be like in 1,000 or 10,000 years from today is likely much greater than the uncertainty associated with the model form, yet this 'future world' uncertainty is not quantifiable or readily bounded. Such sources of uncertainty must be discussed qualitatively rather than being quantitatively modeled.

### 3.4.5 Non-Cancer Toxicity Endpoints

DU waste (and potentially other compounds) associated with the Clive facility can be associated with toxicological risks that are independent of radioactive properties. EPA has evaluated available dose response information for many chemicals and has published this information in the form of toxicity values and accompanying information. Potential health effects related to intake of chemicals is assessed by means of slope factors for suspected carcinogens, and reference doses (RfDs) for noncarcinogenic effects of chemicals. Unlike carcinogenic agents, EPA typically views toxicants with non-cancer effects as having thresholds; i.e., levels below which effects would be unlikely. RfDs essentially amount to such thresholds, usually with several layers of 'safety' factors added.

A limited evaluation of the effect of science policy uncertainty in the value of the uranium oral RfD on chemical hazard results is included in this assessment. The modeling process is very similar to that conducted for radionuclides, other than kidney toxicity (as opposed to radiation dose) of DU will be evaluated, and the toxicity of DU will not change over time (as radioactive decay is not important in this context).Oral toxicity criteria for uranium are published by EPA in relation to the Superfund program (EPA, 2011) and by EPA's Office of Water in relation to drinking water standards (EPA, 2000). There is a five-fold difference between these criteria, and both will be employed in the assessment of uranium toxicity to determine the sensitivity of uranium health effect results to differences in these recommended toxicity criteria for uranium.

A discrete distribution is used to represent the uranium oral RfD based on current EPA science policy associated with EPA’s Superfund Program and Office of Water. A uranium oral RfD of 0.0006 mg/kg-day is associated with the derivation of the final uranium drinking water maximum contaminant level (MCL) as defined on page 76713 of Federal Register, Volume 65, No. 236, December 7, 2000 (Section I.D.2d). A uranium oral RfD of 0.003 mg/kg-day for soluble salts of uranium is published in the Integrated Risk Information System (IRIS) supporting the Superfund Program. A 50/50 probability is assigned to these oral RfDs to determine in the

Sensitivity Analysis whether selecting one or the other of these published values is a significant contributor to uncertainty in the uranium Hazard Index in any exposure scenario.

# 4.0 Equations and Parameters of the Exposure-Dose Container

## 4.1 Organization

The implementation of the exposure and dose calculations, and associated results, are organized within different subcontainers in the ED container. A description of the main subcontainers and their contents are described below:

- Environmental Concentrations: Concentrations of species in various environmental media developed in the Contaminant Transport (CT) component of the PA model are tracked here. These elements are the link between the CT and ED components of the PA model, and take the form of GoldSim vectors defined by the array Species. Environmental concentrations are subsequently defined as two-dimensional matrices with the addition of arrays for different receptor groups in order to track doses for multiple individuals to tally a population dose.
- Behavioral Parameters: Input parameter values related to human activities and behaviors for the Rancher, Sport OHVer, and Hunter exposure scenarios. With few exceptions, these parameters are defined within an 'inner-loop' container that has a separate internal timestep so that they can be sampled on an annual basis regardless of the timestep length of the CT model.
- DCFs: Dose conversion factors for radionuclides are grouped in a subcontainer outside the inner-loop container..
- Dose Calculations: A series of subcontainers are defined within the inner-loop container for calculation of TEDE related to inhalation, ingestion, and external radiation exposures for the Rancher, Sport OHVer, and Hunter exposure scenarios. A container for off-site receptor doses is also provided. Screening-level dose calculations for ingestion of edible plant materials gathered on the waste disposal cell, and ingestion of standing surface water, are grouped in a subcontainer outside the inner-loop container.
- Uranium Hazard: A subcontainer within the inner-loop container holding calculations for systemic toxicity (hazard) related to the nonradiological effects of uranium.

In terms of parameter definitions, GoldSim uses a variety of methods, including deterministic values, scalars, time series data, and “stochastics”, which are user-defined statistical distributions. Parameter distributions employed in the PA model reflect a mixture of site- and receptor-specific data, information modeled in 'upstream' portions of the PA model, literature information, and subjective judgment; as appropriate.

## 4.2 Environmental Concentrations

The principal link between the CT component and the ED component of the PA model are concentrations of contaminants in different environmental media. Major environmental media evaluated in the ED container include:

- Soil. There are several soil concentration terms that are used in the ED container. The contaminant transport portion of the PA model employs a homogenized waste source term and simulates transport over time to produce estimates of soil concentrations for the embankment top slope and the embankment side slopes. The principal soil term in the ED container is the area-weighted average concentration in the top layer of both the top slope and side slope of the disposal cap. This is the disposal cap soil concentration. Contaminant concentrations in these soils, plus possible contribution from lower soil layers and even the disposed waste itself, are used to calculate soil exposure concentrations for the embankment. Embankment soil concentrations are defined as the area-averaged soil concentrations of the disposal cap and of one or more gullies and fans that may develop in the future. Finally, particle resuspension and deposition models are used to calculate area-averaged soil concentrations off-site air dispersion area based upon the embankment soil concentrations.

  Area-averaged soil concentrations for the embankment and the off-site air dispersion area are employed because there is no basis for specifying greater or lesser individual exposure intensity as a function of location within these regions. Individuals are presumed to be exposed at random in these areas, and an area-averaged exposure concentration reflects this presumed behavior.

  The human exposure area surrounding the Clive site is where the Ranchers, Sport OHVers, and Hunters identified as likely receptor populations conduct their activities. The maximum size of this area is the approximate area between I-80 and the Utah Test and Training Range (UTTR) in an east-west orientation, and the Cedar Mountain foothills and salt/mud flats in a north-south orientation. The minimum size of this area is the approximate minimum size of the four current grazing leases in the vicinity of the Clive facility. Because the maximum area is roughly equivalent to the largest of the four current grazing leases, the human exposure area and the size of the area over which cattle may graze are equivalent.

- Air. Air concentrations of gaseous and particulate contaminants in the atmosphere are calculated using the AERMOD atmospheric dispersion model for breathing-zone air above the embankment and above the off-site dispersion area. Off-site air concentrations are also calculated at the specific exposure locations described in Section 3.2.3. These calculations are documented in the *Atmospheric Transport Modeling* white paper (Appendix 8). To evaluate the impacts of dust generated during off-highway vehicle (OHV) use, an adjustment factor for particulate air concentrations is used based upon dust generation data collected by EPA Region 9 for OHV users wearing personal air monitors in a recreational area in California (EPA, 2008).

- Game. Contaminant concentrations in the meat of game animals that incorporate the embankment and nearby areas as part of their home range. Based upon

communications with BLM, pronghorn are modeled as the most likely game species of interest to future Hunters. Contaminant concentrations in game tissue are modeled as a function of ingestion of browse plants, standing surface water, and soil inadvertently ingested while browsing.

- Beef. Contaminant concentrations in beef from cattle that incorporate the embankment and nearby areas as part of their range. Similar to game tissue concentrations, beef concentrations are related to plants, surface water, and soil. The number of cattle grazing in impacted areas is assumed to be sufficient to provide ranchers with beef commensurate with the specified intake rates.
- Plants. Wet weight contaminant concentrations in plant tissues. These concentrations are used as an interim step in the calculation of tissue concentrations in cattle and game and are calculated assuming equilibrium with soil defined by element-specific plant-soil concentration ratios. They are also used for screening-level calculations to determine if potential direct human exposures by plant ingestion may be of concern.
- Surface Water. Contaminant concentrations in standing surface water in the air dispersion area. Water concentrations are calculated assuming equilibrium with soil, as defined by element-specific soil-water partition coefficients. These water concentrations are used as an interim step in the calculation of tissue concentrations in cattle and game. They are also used for screening-level calculations to determine if potential direct human exposures by surface water ingestion may be of concern.

Groundwater is not an exposure medium *per se*, because the aquifer below the Clive facility is too saline to be used as a drinking water source, and so is classified by the State of Utah as Class IV (nonpotable) in the ground water quality discharge permit for the Clive facility. However, the permit also states that concentrations of contaminants in groundwater will nevertheless be compared to State of Utah GWPLs.

## 4.3 Exposure Parameters

The basis of the deterministic values and/or statistical distributions for each of the ED equation parameters is discussed in the *Model Parameters* white paper (Appendix 16), the attached Appendix I, and the spreadsheet *Dose Assessment Appendix II*. A major source of exposure parameter values is the 2009 update to the EPA *Exposure Factors Handbook* (EPA, 2009a). Although this reference exists as an external review draft, it is much more current and extensive than the 1997 version, and much more distributional information is included. For physiological variables in particular, the primary studies that EPA employed as the basis of recommendations in EPA (2009a) were also reviewed.

Three non-residential human receptor scenarios (Rancher, Sport OHV recreationist, and Hunter recreationist) are defined, each with its own set of exposure parameter values but with similar computational exposure models. Exposure parameters that pertain to inter-individual population variability have been assigned to the "inner loop" of the 2D Monte Carlo simulation. These parameters pertain to physiological characteristics, the fraction of time an individual spends on or near the site, and the number of receptors present at the site. These categorizations of inner or outer loop are noted in Section 1 and discussed in Section 3.4.1.

Exposure parameters related to inter-individually varying population characteristics, and to the number of receptors within the exposure area, are defined within an “inner-loop” sub-container in the ED model. This sub-container has an annual time step so that the stochastic parameters relating to the number of individuals appearing in the exposure area, and the inter-individual characteristics of these individuals, are sampled annually. This sub-container is the "inner loop" of the 2-dimensional Monte Carlo simulation.

The remainder of the exposure parameters, which include the exposure concentrations in environmental media, the DCFs, and a few other parameters, are defined by uncertainty distributions that apply to each individual in the population over the entire 10,000-yr performance period. These parameters, and all components of the contaminant transport model that produce estimates of exposure concentrations over time, are in the "outer loop" of the 2-dimensional Monte Carlo simulation. The uncertainty distributions for stochastic parameters in the outer loop outside this sub-container are sampled only once at the beginning of each model realization.

In the 2-dimensional model, it is assumed that uncertainties are independent for each member of the ranching and recreational scenario populations. The fraction of time that each individual spends on the disposal cell or in the adjacent off-site area is variable. Because the processes that lead to concentration terms in these two areas are different, they have different uncertainty characteristics. This results in independence in the uncertainties of the individual annual dose results.

The inhalation rate distributions activities are specified according to exertion level as heavy, moderate, light, sedentary, and sleeping. For each exertion level, EPA (2009a) provides information for breathing (ventilation) rate and associated fraction of daily time spent at that level. In the absence of scenario-specific information, the fraction of daily time spent at each exertion level for the general population described in EPA (2009a) has been applied to ranching and recreation receptors. Stochastic distributions for the inhalation rates, and also for meat ingestion rates, are tied to the age and (for inhalation rate) gender of an individual receptor, and are specified as a linear function of their body weight as described in EPA (2009a; 2009b). An adult between the ages of 16 and 60 is defined for the ranching and recreation receptor groups.

The behavioral exposure parameters defined in the inner-loop sub-container relate primarily to the fraction of daily and yearly time spent by receptors in the exposure area generally, and within the exposure area the fractional time spent on the embankment versus other locations. Based upon discussion with BLM, Ranchers are assumed to work within a ranching lease during the day and may also camp overnight. Both Sport OHVers riders and Hunters may visit the area for either a day trip or an overnight trip.

## 4.4 DCFs

The TEDE is not an effect *per se*, but rather a measure of radiation dose absorbed by a tissue. The DCFs used in the ED model account for the biological effectiveness of the radiation (e.g., alpha particles, photons) in causing cellular damage in different tissues, as well as the sensitivity of different tissues to the effects of ionizing radiation. For external dose, this “effective dose” is calculated. For internal dose, the committed effective dose is calculated, which accounts for continued dose over time from radionuclides retained in the body. Distribution development for

one source of uncertainties inherent in DCFs (i.e., associated with REFs) is described in Section 3.4.3.

Section 3.3.7 of NUREG-1573 (NRC, 2000) discusses modeling of radiation dose, including internal and external dosimetry. NRC (2000) notes that the performance objectives set forth in Section 61.41 of Title 10 CFR 61.41 (CFR, 2007) are based upon ICRP 2 dose assessment methods, which pre-date the development of TEDE methodology. NRC recommends the use of current ICRP dosimetry employing TEDE methods in lieu of calculation of individual organ doses. The internal and external DCFs used in the ED model were obtained from the electronic database accompanying FGR 13 (EPA, 1999), available online at http://ordose.ornl.gov/downloads.html and also provided in the spreadsheet *Dose Assessment Appendix II*. The DCFs for all species, as well as the individual short-lived progeny of these parent nuclides, were developed using appropriate decay chains and branching fractions as described in the CSM and documented in the electronic attachment.

The DCF for radon-222 and progeny was derived from recommendations provided in an ICRP draft report for consultation (ICRP, 2009). A range of 3 - 6 mSv-$m^3$/mJ-hr is given for the radon-222 DCF, calculated using ICRP's Human Respiratory Tract Model. The main sources of uncertainty related to this range are the activity size distribution of aerosols for radon progeny, and the breathing rates (ICRP 2009; Appendix B, paragraph B 6).

In paragraph B 11 of Appendix B to ICRP (2009), the inhalation rate for a "standard worker" associated with the upper-end DCF estimate of 6 mSv-$m^3$/mJ-hr is given as 1.2 $m^3$/hr. ICRP states,

> *For typical aerosol conditions in home and mines the effective dose is about 3.7 mSv-$m^3$/mJ-hr. . . However, assuming the same aerosol conditions as for a home but with a breathing rate for a standard worker (1.2 $m^3$/hr) the effective dose increases from 3.7 to 6 mSv-$m^3$/mJ-hr.*

This indicates that approximately 75% of the range of 3 - 6 mSv-$m^3$/mJ-hr given for the Rn-222 DCF may be related to inhalation rate. Based upon this observation, a breathing rate normalized radon-222 DCF was calculated for use in the ED model. The units for alpha energy (mJ) were converted to an equivalent activity (Bq) for radon-222 according to units definitions in the glossary of ICRP (2009).

A radon-222 DCF of $2.8 \times 10^{-8}$ Sv/Bq was calculated as:

$$\text{Radon-222 DCF} = (0.006 \text{ Sv-m3/mJ-hr} \times 5.56 \times 10\text{-}6 \text{ mJ/Bq}) / 1.2 \text{ m}^3\text{/hr}$$

Note that the REFs discussed earlier are not applicable to radon, as the DCF was estimated in a different fashion than the other species.

## 4.5 PDCFs

A PDCF is an equation that combines Exposure Parameter values and DCFs, as described in Section 3.3.7.2 of NRC (2000). PDCFs are combined with estimates of radionuclide concentrations in exposure media to calculate a TEDE. PDCF equations for each exposure route are described in subsections below.

### 4.5.1 Inhalation PDCF Equations

PDCF for inhalation of particulates and gases

$$PDCF\_Inh\ (\text{Sv}\cdot\text{m}^3/\text{Bq}\cdot\text{yr}) = DCF\_Inh \times InhalationRate \times EF \times ET \quad (1)$$

where

*DCF_Inh* is the inhalation DCF (Sv/Bq)

*InhalationRate* is the activity-weighted inhalation rate ($m^3$/hr)

*EF* is the yearly exposure frequency (d/yr), and

*ET* is the total daily exposure time (hr/d).

and

$$InhalationRate\ (\text{m}^3/\text{hr}) = \sum_i (Inhal\_act_i \times ET\_frac_i) \quad (2)$$

where

$Inhal\_act_i$ is the inhalation rate for activity level *i* ($m^3$/hr), and

$ET\_frac_i$ is the fraction of daily exposure time for activity level *i* (-)

Activity levels (*i*) for which population-average breathing rates and daily exposure times are defined include sleeping, sedentary activity, light activity, medium activity, and heavy activity. Breathing rates are body weight adjusted. Population distributions of both breathing rates and daily exposure times at different activity levels are defined as functions of age and gender, as described in EPA (2009a).

### 4.5.2 External PDCF Equations

PDCF for external radiation from soil

$$PDCF\_Ext\_Soil\ (\text{Sv}\cdot\text{g}/\text{Bq}\cdot\text{yr}) = DCF\_Ext \times EF \times ET \times \rho_b \times CF_1 \quad (3)$$

where

*DCF_Ext* is the external DCF for a 3-dimensional soil source ($Sv\cdot m^3/Bq\cdot s$)

*EF* is the yearly exposure frequency (d/yr)

*ET* is the total daily exposure time (hr/d)

$\rho_b$ is the bulk soil density ($g/m^3$), and

$CF_1$ is a unit conversion factor (3600 s/hr)

PDCF for external radiation from immersion in air

$$PDCF\_Imm\ (\text{Sv}\cdot\text{m}^3/\text{Bq}\cdot\text{yr}) = DCF\_Imm \times EF \times ET \times CF_1 \quad (4)$$

where

*DCF_Imm* is the external DCF for air immersion (Sv·m3/Bq·s)

*EF* is the yearly exposure frequency (d/yr)

*ET* is the total daily exposure time (hr/d), and

*CF1* is a unit conversion factor (3600 s/hr)

### 4.5.3 Ingestion PDCF Equations

PDCF for inadvertent ingestion of soil

$$PDCF\_Ing\_Soil\ (\text{Sv}\cdot\text{g/Bq}\cdot\text{yr}) = DCF\_Ing \times SoilIngRate \times EF \times CF_2 \quad (5)$$

where

*DCF_Ing* is the ingestion DCF (Sv/Bq)

*SoilIngRate* is the daily soil ingestion rate (mg/day)

*EF* is the yearly exposure frequency (d/yr), and

*CF2* is a unit conversion factor (0.001 g/mg).

PDCF for ingestion of game meat or beef

$$PDCF\_Ing\_Meat\ (\text{Sv}\cdot\text{g/Bq}\cdot\text{yr}) = DCF\_Ing \times MeatConsumpRate \times (1 - Prep\_loss) \times (1 - PostCook\_loss) \times EF\_food \quad (6)$$

where

*DCF_Ing* is the ingestion DCF (Sv/Bq)

*MeatConsumpRate* is the daily consumption rate of beef or game meat (g/kg body weight/d)

*Prep_loss* is the fractional preparation and cooking loss of consumed meat related to dripping and volatile losses during cooking (-)

*PostCook_loss* is the fractional post-cooking loss of consumed meat related to trimming, bones, scraps, etc (-)

*EF_food* is the intrinsic exposure frequency assumed in the time-averaged ingestion rate data (d/yr)

PDCF for plant ingestion (screening calculation)

$$PDCF\_Ing\_Plant\ (\text{Sv}\cdot\text{g/Bq}\cdot\text{yr}) = DCF\_Ing \times PlantIngRate \quad (7)$$

where

*DCF_Ing* is the ingestion DCF (Sv/Bq, and

*PlantConsumpRate* is the yearly consumption rate of wild plants (g/yr)

PDCF for water ingestion (screening calculation)

$$PDCF\_Ing\_Water\ (\text{Sv}\cdot\text{g/Bq}\cdot\text{yr}) = DCF\_Ing \times WaterIngRate \times WatDens \quad (8)$$

where

*DCF_Ing* is the ingestion DCF (Sv/Bq,

*WaterConsumpRate* is the yearly consumption rate of standing water (L/yr), and

*WatDens* is the density of water (g/L)

## 4.6 TEDE

The calculation of dose, represented here by TEDE, is the product of a PDCF and the exposure concentration. Separate soil concentrations are developed in the contaminant transport model for the disposal cap and the off-site area impacted by deposition of wind-dispersed particles. Particulate air concentrations, which are related to resuspension of soil, and concentrations of gas-phase radionuclides in air, are also calculated separately for these three exposure areas. Other exposure concentrations used in the dose model include radionuclide concentrations in animal tissue, as well as plant tissue and standing surface water in screening calculations.

All TEDE calculations reference the PA model element describing the time after site closure when institutional controls fail and a receptor can gain access to the site. If this time has not been reached in the model realization, ranching and recreation doses are assigned a zero value.

Note that potential embankment gullies are modeled in a preliminary manner in the PA model to evaluate possible consequences given the current waste disposal configuration. Gully formation can be 'switched' on or off by the model user.

### 4.6.1 Inhalation TEDE Equations

Gas and particulate inhalation TEDE results (mSv/yr) are vectors dimensioned by Species in the PA model related to the inhalation PDCFs. Concentrations of respirable particles and gas-phase radionuclides in air are calculated by methods described in the *Atmospheric Transport Modeling* white paper (Appendix 8). The inhalation TEDE equation for particulate inhalation is:

$$TEDE\_Inh\ (\text{mSv/yr}) = PDCF\_Inh \times C_{air} \tag{9}$$

where

*PDCF_Inh* is the inhalation PDCF ($Sv \cdot m^3/Bq \cdot yr$), and

$C_{air}$ is the spatially-averaged air concentration ($Bq/m^3$)

Exposure concentrations on the embankment are calculated in an area-weighted manner. This calculation presumes that exposures across the embankment occur in a random manner. Air concentrations above the embankment are calculated as:

$$C_{embnk}\ (Bq/m^3) = \{([C_{cap} \times A_{cap} + C_{gullies} \times A_{gullies}] / [A_{cap} + A_{gullies}]) \tag{10}$$

where

$C_{embnk}$ is the air concentration above the embankment ($Bq/m^3$)

$C_{cap}$ is the air concentration above the disposal cap ($Bq/m^3$)

$A_{cap}$ is the area of the embankment cap ($m^2$)

$C_{gullies}$ is the air concentration above the gullies and associated fans ($Bq/m^3$)

$A_{gullies}$ is the surface area of the gullies and associated fans ($m^2$)

The terms $C_{gullies}$ and $A_{gullies}$ are calculated using a model for possible erosive effects of precipitation subsequent to gully initiation due to OHV activity, grazing animals, or other processes.

With respect to ranching and recreation exposure, there are two concentrations terms to address: a concentration term for the embankment (per Equation 10) and a concentration term for the off-site air dispersion area. A weighted exposure concentration for particulates in ambient air is calculated for these two concentration terms as follows:

$$C_{air\text{-}dust}\ (\mathrm{Bq/m^3}) = \{\ OHV\_timefrac \times OHV\_dust \times (C_{embnk} \times ET\_frac_{embnk}) + (C_{a\text{-}disp} \times [1 - ET\_frac_{embnk}]\ \} + \{\ [\ 1 - OHV\_timefrac] \times (C_{embnk} \times ET\_frac_{embnk}) + (C_{a\text{-}disp} \times [1 - ET\_frac_{embnk}]\ \} \quad (11)$$

where

*OHV_timefrac* is the fraction of exposure time spent OHVing (-)

*OHV_dust* is the off-highway vehicle dust factor, used to account for the contribution of mechanical dust creation (-)

$C_{embnk}$ is the air concentration above the embankment ($\mathrm{Bq/m^3}$)

$ET\_frac_{embnk}$ is the fraction of total daily exposure time spent on the embankment (-)

$C_{a\text{-}disp}$ is the air concentration above the air dispersion area ($\mathrm{Bq/m^3}$)

For particulates, $C_{embnk}$ and $C_{a\text{-}disp}$ are calculated using a particle erosion model, which calculates the amount of dust released from the ground surface, and the AERMOD air dispersion model (see the *Atmospheric Transport Modeling* white paper, Appendix 8). Particle erosion is assessed as a function of both wind and mechanical disturbance from the use of OHVs, but the mechanical dust creation factor is applied as a multiplier to the baseline (wind-derived) dust concentration. The AERMOD air dispersion model is used to estimate particulate deposition in the offsite air dispersion area as well as breathing zone concentrations of respirable particles above contaminated soil.

For radon and other gas-phase radionuclides, $C_{embnk}$ and $C_{a\text{-}disp}$ are calculated using AERMOD (see the *Atmospheric Transport Modeling* white paper, Appendix 8) based upon the embankment surface flux computed in the PA model. The air dispersion area is not a definite region with respect to particle definition, because it's size is defined by the size of the receptor exposure area, which varies as described in Section 4.2. Based upon AERMOD calculations, a protective estimate of respirable particle deposition beyond the embankment is assigned to this area. Radon air concentrations in the off-site air dispersion area are calculated as the average across the entire area.

Because mechanical dust generation by OHVs is not an issue for calculating air concentrations of radon and other gas-phase radionuclides, Equation 11 reduces to:

$$C_{air\text{-}gas}\ (\mathrm{Bq/m^3}) = (C_{embnk} \times ET\_frac_{embnk}) + (C_{a\text{-}disp} \times [1 - ET\_frac_{embnk}]) \quad (12)$$

The current version of the PA model does not fully integrate gully formation into the physical model of the embankment. Therefore, Radon air concentrations in the gully are modeled from estimated radium-226 surface soil concentrations on the gully 'floor'. The contribution of radon from disposed waste below this surface soil layer is presently accounted for. Also, the influence

of gully walls on radon air concentrations within the gully has not been modeled. For these reasons, gully radon exposures may be underestimated.

### 4.6.2 External Radiation TEDE Equations

Soil and air immersion external dose results (mSv/yr) are vectors dimensioned by Species in the PA model related to the external PDCFs.

The air immersion external dose equation is:

$$TEDE\_Imm \text{ (mSv/yr)} = PDCF\_Imm \times C_{air} \quad (13)$$

where

$PDCF\_Imm$ is the immersion PDCF ($Sv \cdot m^3/Bq \cdot yr$), and

$C_{air}$ is the spatially-averaged air concentration ($Bq/m^3$)

The derivation of $C_{air}$ for air immersion is identical to that described in Equations 10, 11 and 12.

The soil external dose equation is:

$$TEDE\_Ext\_Soil \text{ (mSv/yr)} = PDCF\_Ext\_Soil \times C_{soil} \quad (14)$$

where

$PDCF\_Ext\_Soil$ is the soil ingestion PDCF ($Sv \cdot g/Bq \cdot yr$), and

$C_{soil}$ is the spatially-averaged soil concentration (Bq/g)

Similar to Equation 8, soil concentrations on the embankment are calculated as:

$$C_{embnk} \text{ (Bq/g)} = \{([C_{cap} \times A_{cap} + C_{gullies} \times A_{gullies}] / [A_{cap} + A_{gullies}]) \quad (15)$$

where

$C_{embnk}$ is the embankment soil concentration (Bq/g)

$C_{cap}$ is the disposal cap soil concentration (Bq/g)

$A_{cap}$ is the area of the disposal cap ($m^2$)

$C_{gullies}$ is the soil concentration of the gullies and associated fans (Bq/g)

$A_{gullies}$ is the surface area of the gullies and associated fans ($m^2$)

Analogous to Equation 12, a weighted exposure concentration for embankment and air dispersion area soil is calculated as follows:

$$C_{soil} \text{ (Bq/g)} = (C_{embnk} \times ET\_frac_{embnk}) + (C_{a\text{-}disp} \times [1 - ET\_frac_{embnk}]) \quad (16)$$

where

$ET\_frac_{embnk}$ is the fraction of total daily exposure time spent on the embankment (-)

$C_{a\text{-}disp}$ is the soil concentration for the air dispersion area (Bq/g)

### 4.6.3 Ingestion TEDE Equations

Inadvertent soil ingestion (i.e., via soil on hands, food, etc.) and meat ingestion dose results (mSv/yr) are vectors dimensioned by Species in the PA model related to the ingestion PDCFs.

The soil inadvertent ingestion dose equation is:

$$TEDE\_Ing\_Soil \text{ (mSv/yr)} = PDCF\_Ing\_Soil \times C_{soil} \quad (17)$$

where

*PDCF_Ext_Soil* is the soil ingestion PDCF (Sv·g/Bq·yr), and

$C_{soil}$ is the spatially-averaged soil concentration (Bq/g)

($C_{soil}$ is calculated according to Equation 16.)

The meat ingestion dose equation is:

$$TEDE\_Ing\_Meat \text{ (mSv/yr)} = PDCF\_Ing\_Meat \times C_{meat} \quad (18)$$

where

*PDCF_Ext_Soil* is the soil ingestion PDCF (Sv·g/Bq·yr), and

$C_{meat}$ is the concentration in beef or game meat (Bq/g)

The calculation of $C_{meat}$ is based upon grazing models for beef cattle and pronghorn and uses as inputs the soil concentrations $C_{embnk}$ and $C_{a\text{-}disp}$. Both beef cattle and pronghorn may be exposed to soil contamination by direct soil ingestion while grazing, by ingestion of browse plants growing in contaminated soil, and by ingestion of standing water on contaminated soil.

Radionuclide concentrations in beef and game tissue are calculated based upon three animal exposure pathways: direct ingestion of soil while browsing, ingestion of plants growing in contaminated soils, and drinking standing surface water. Cattle and pronghorn are assumed to graze randomly across the entire range area. Hence, exposure to radionuclides in the embankment and air dispersion areas is based upon the relative size of these areas.

For soil, exposure concentrations for cattle are calculated as:

$$C_{soil\text{-}cattle} \text{ (Bq/g)} = ([C_{embnk} \times A_{embnk}] + [C_{a\text{-}disp} \times A_{a\text{-}disp}]) / A_{range\text{-}cattle} \quad (19)$$

where

$C_{embnk}$ is the embankment soil concentration (Bq/g)

$A_{embnk}$ is the area of the embankment ($m^2$)

$C_{a\text{-}disp}$ is the soil concentration for the air dispersion area (Bq/g)

$A_{a\text{-}disp}$ is the surface area of the air dispersion area ($m^2$), and

$A_{range\text{-}cattle}$ is the size of the cattle range area (m2)

Soil radionuclide exposure concentrations for pronghorn are calculated in an identical manner, substituting the size of the pronghorn grazing area.

Equation 19 is also used to calculate exposure concentrations in browse plants for cattle and pronghorn. However, plant concentrations on the disposal cap are based upon uptake of

contamination across the entire root depth profile of the plants. Different types of plants (differentiated by root depth distributions, biomass, and leaf litter production) are employed in the Contaminant Transport component of the PA model to evaluate transport of radionuclides on the disposal cap. Plant concentrations on the disposal cap are calculated in the contaminant transport portion of the PA model as the weighted average (based upon leaf litter production) of all plants. Soil concentrations in the air dispersion area, and in the gullies and fans, are only calculated for a single surface soil layer. 100% of plant roots are assumed to be situated in this layer.

For standing surface water, exposure concentrations for cattle and pronghorn are calculated for puddles in the air dispersion area. Puddle water concentrations are based upon bulk soil concentrations using element-specific soil water partition coefficients.

Using the exposure concentrations described above, radionuclide concentrations in beef are calculated as:

$$C_{beef}\,(\text{Bq/g}) = TF\_beef \times (C_{plant\text{-}cattle} \times cattle\_forage) + (C_{soil\text{-}cattle} \times cattle\_soil) + (C_{water\text{-}cattle} \times cattle\_water) \quad (20)$$

where

*TF_beef* is the amount of an element taken up into muscle tissue as a function of the daily intake rate of that element by the animal. (Bq/g per Bq/d)

$C_{plant\text{-}cattle}$ is the area-weighted plant concentration on the cap, gullies and fans, and air-dispersion area (Bq/g dry wt)

*cattle_forage* is the dry-weight forage intake rate for browsing cattle (g/day dry wt)

$C_{soil\text{-}cattle}$ is the weighted soil concentration on the embankment and air-dispersion areas (Bq/g)

*cattle_soil* is the soil ingestion rate for browsing cattle (g/day)

$C_{water\text{-}cattle}$ is the water concentration for the puddles in the air-dispersion areas (Bq/g)

*cattle_water* is the water ingestion rate for browsing cattle (g/day)

Concentrations in pronghorn tissue ($C_{game}$) are calculated in a manner analogous to Equation 20, substituting weighted exposure concentrations and intake rates for pronghorn.

Transfer factors (TFs) determine the amount of an element taken up into muscle tissue as a function of the daily intake rate of that element by the animal. The units are expressed as Bq/kg per Bq/d (d/kg). Element-specific beef transfer factors were preferentially obtained from a recent publication of the International Atomic Energy Agency (IAEA, 2010). A report by Pacific Northwest National Laboratory (Staven et al., 2003) was used as a secondary reference. For many elements, these values are reported as a geometric mean and geometric standard deviation. For a subset of elements with only a single reference, an arithmetic mean is provided with no measure of variance. In these cases (actinium, americium, neptunium, protactinium, radium, and technetium), an estimate of variance was produced by taking the average geometric standard deviation for the all other elements excepting plutonium, which was considered an outlier. A summary of the beef TFs with accompanying notes is provided in Table 3.

Distributional form for the values of geometric mean and geometric standard deviation reported in IAEA (2010) was not discussed in this reference. Also, For sample sizes of less than 3, IAEA (2010) values were originally reported as the arithmetic mean and standard deviation. In order to provide a common set of inputs, values obtained from IAEA (2010) and Staven et al. (2003) were processed to conform to an assumed lognormal distribution. Values originally reported as arithmetic mean and standard deviation were transformed to geometric equivalents.

Beef TF data were reported in IAEA (2010) as a geometric mean, geometric standard deviation, minimum, and maximum. The geometric standard deviations are greater than 2 in nearly every case, suggested high right-skewness in the data, and the minimum and maximum were consistent with samples from a lognormal distribution. In order to establish a distribution for the mean, a parametric bootstrap approach was taken [Efron 1998], simulating bootstrap samples from the lognormal distribution using the maximum likelihood estimates of the lognormal parameters. A lognormal distribution was then fit to the resulting bootstrap simulations of the mean, since some right-skewness was still present in the sampling distribution.

**Table 3. Beef transfer factors (Bq/kg per Bq/d)**

| Element | Sample size | Geometric Mean | Geometric Std. Dev. | Notes |
|---|---|---|---|---|
| Actinium | 1 | 0.0004 | generic* | Mean based upon Staven et al. (2003; table 2-6, p. 2.7); no value in IAEA (2010). Geometric standard deviation based upon 6 surrogate elements. |
| Americium | 1 | 0.0005 | generic* | Geometric standard deviation based upon 6 surrogate elements. (IAEA, 2010; table 30, p. 93) |
| Cesium | 58 | 0.032 | 1.15 | Based upon values provided in IAEA (2010; table 30, p. 93)). |
| Iodine | 5 | 0.0107 | 1.85 | Based upon values provided in IAEA (2010; table 30, p. 93). |
| Neptunium | 1 | 0.001 | generic* | Mean based upon Staven et al. (2003; table 2.6, p. 2.7); no value in IAEA 2010. Geometric standard deviation based upon surrogate elements. |
| Protactinium | 1 | 0.0005 | generic* | Americium (IAEA 2010 value) used as a surrogate based upon Staven et al. (2003; table 2.6, p. 2.7). |
| Lead | 5 | 0.000952 | 1.59 | Based upon values provided in IAEA (2010; table 30, p. 93). |
| Plutonium | 5 | 0.0000128 | 7.42 | Based upon values provided in IAEA (2010; table 30, p. 93). |
| Radium | 1 | 0.0017 | generic* | Geometric standard deviation based upon surrogate elements. (IAEA 2010; table 30, p. 93) |

| Element | Sample size | Geometric Mean | Geometric Std. Dev. | Notes |
|---|---|---|---|---|
| Radon | -- | arbitrarily small value | 1 | Radon gas is inert and has effectively no potential to establish an equilibrium in animal tissue. |
| Strontium | 35 | 0.00223 | 1.26 | Based upon values provided in IAEA (2010; table 30, p. 93). |
| Technetium | 1 | 0.0001 | generic* | Mean based upon Staven et al. (2003; table 2.6, p. 2.7); no value in IAEA 2010. Geometric standard deviation based upon surrogate elements. |
| Thorium | 6 | 0.000355 | 1.68 | Based upon values provided in IAEA (2010; table 30, p. 93). |
| Uranium | 3 | 0.000421 | 1.32 | Based upon values provided in IAEA (2010; table 30, p. 93). |

* A generic GSD for these elements is 1.475

# 5.0 References


BLM (United States Bureau of Land Management) 2010. *Rangeland Administration System*. United States Bureau of Land Management, www.blm.gov/ras.

Bogen K.T., Cullen A.C., Frey H.C., Price P. 2009. Probabilistic exposure analysis for chemical risk characterization. *Toxicological Sciences* 109(1): 4-17.

Brenner D.J., Doll R., Goodhead D.T., et al. 2003. Cancer risks attributable to low doses of ionizing radiation: Assessing what we really know. *Proc. Natl Acad Sci* 100(24): 13761-6.

Burr S.W., Smith J.W., Reiter D., et al. 2008. *Recreational Off-Highway Vehicle Use on Public Lands in Utah*. Utah State University, Institute for Outdoor Recreation and Tourism, Logan UT.

CFR (Code of Federal Regulations). 1994. *National Oil and Hazardous Substances Pollution Contingency Plan*, Code of Federal Regulations 40 CFR 300, US Government Printing Office, Washington, DC.

CFR. 2007. *Licensing Requirements for Land Disposal of Radioactive Waste*, Code of Federal Regulations Title 10, Part 61 (10 CFR 61), US Government Printing Office, Washington, DC.

Davis S., Mirick D.K. 2006. Soil ingestion in children and adults in the same family. *Journal of Exposure Science and Environmental Epidemiology*, 16: 63-75.

Efron B.,Tibshirani R.J. 1998. *Introduction to the Bootstrap*, CRC Press, Boca Raton, FL.

EPA (United States Environmental Protection Agency). 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion*, EPA 520/1-88-020, Federal Guidance Report (FGR) 11, US Environmental Protection Agency, Washington, DC.

EPA 1999. *Cancer Risk Coefficients for Environmental Exposure to Radionuclides*. Federal Guidance Report No. 13, United States Environmental Protection Agency and Oak Ridge National Laboratory, EPA 402-R-99-001, Oak Ridge TN.

EPA 2007 (Pawel et al.) *Uncertainties in Cancer Risk Coefficients for Environmental Exposure to Radionuclides: An Uncertainty Analysis for Risk Coefficients Reported in Federal Guidance Report No. 13*, United States Environmental Protection Agency and Oak Ridge National Laboratory, ORNL/TM-2006/583, Oak Ridge TN.

EPA, 2011a. *Integrated Risk Information System* (IRIS), Office of Research and Development and National Center for Environmental Assessment, Electronic Database. Available online at: http://www.epa.gov/iris/.

EPA. 1989. *Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A)*, Interim Final, Office of Emergency and Remedial Response, US Environmental Protection Agency, Washington, DC.

EPA. 1993a. *External Exposure to Radionuclides in Air, Water, and Soil*, EPA 402-R-93-81, Federal Guidance Report (FGR) 12, US Environmental Protection Agency, Washington, DC.

EPA. 1993b. *Wildlife Exposure Factors Handbook*. Vol. I. EPA/600/R-93/187, Office of Research and Development, US Environmental Protection Agency, Washington, DC.

EPA. 1997a, *Establishment of Cleanup Levels for CERCLA Sites with Radioactive Contamination*, Memorandum from Stephen D. Luftig, Director, Office of Emergency and Remedial Response, and Larry Weinstock, Acting Director, Office of Radiation and Indoor Air, to addressees, OSWER No. 9200.4 18, US Environmental Protection Agency, Washington, DC.

EPA. 1997b. *Exposure Factors Handbook*. Office of Research and Development. US Environmental Protection Agency, Washington, DC.

EPA. 1999. *Cancer Risk Coefficients for Environmental Exposure to Radionuclides: Updates and Supplements*, EPA 402-R-99-001, FGR 13, US Environmental Protection Agency, Washington, DC.

EPA. 2000. *National Primary Drinking Water Regulations; Radionuclides; Final Rule*, Federal Register: December 7, 2000 (Volume 65, Number 236)

EPA. 2001, *Risk Assessment Guidance for Superfund (RAGS), Volume III: Process for Conducting Probabilistic Risk Assessment (Part A)*, EPA 540-R-02-002, OSWER 9285.7-45, Office of Emergency and Remedial Response, US Environmental Protection Agency, Washington, DC.

EPA. 2005, *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Final*, EPA/530-R-05-006, Office of Solid Waste and Emergency Response, US Environmental Protection Agency, Washington, DC.

EPA. 2008, *Clear Creek Management Area Asbestos Exposure and Human Health Risk Assessment*, US Environmental Protection Agency, Region 9, San Francisco, CA.

EPA. 2009a. *Exposure Factors Handbook*: 2009 Update. External Review Draft, July 2009. Office of Research and Development, National Center for Environmental Assessment, US Environmental Protection Agency, Washington, DC.

EPA. 2009b. *Metabolically Derived Human Ventilation Rates*. May 2009. Office of Research and Development, National Center for Environmental Assessment, US Environmental Protection Agency, Washington, DC.

Filipsson A.F., Sand S., Nilsson J., et al. 2003. The benchmark dose method- review of available models, and recommendations for application in health risk assessment. *Crit Rev Toxicol* 33:505-542.

GTG (GoldSim Technology Group), 2010. *GoldSim: Monte Carlo Simulation Software for Decision and Risk Analysis*, http://www.goldsim.com

Hamby D.M. 1999. Uncertainty of the tritium dose conversion factor. *Health Phys* 77(3): 291-7.

Harvey R.P., Hamby D.M., Palmer T.S., 2006. Uncertainty of the thyroid dose conversion factor for inhalation intakes of I-131 and its parametric uncertainty. *Radiat Prot Dosimetry* 118(3):296-306.

Hendee W.R., Edwards F.M. 1986. ALARA and an integrated approach to radiation protection. S*eminars in Nuclear Medicine* 16:142-150.

Huffman B. 2004. *Antilocapra americana. Pronghorn*. Ungulate Fact Sheet. http://www.ultimateungulate.com/Artiodactyla/Antilocapra_americana.html.

IAEA (International Atomic Energy Agency). 2010. *Handbook of Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments*, International Atomic Energy Agency Technical Report Series No. 472, Vienna.

ICRP, 1977. *Radiation Protection.* International Commission of Radiological Protection Publication No. 26. Permagon, NY.

ICRP. 1979. *Limits for Intakes of Radionuclides by Workers*, International Commission on Radiological Protection Publication No. 30, *Ann. ICRP* 2(3-4).

ICRP. 1983. *Cost-Benefit Analysis in the Optimization of Radiation Protection.* International Commission of Radiological Protection Publication No. 37. Permagon, NY.

ICRP. 1989. *Age-Dependent Doses from Intakes of Radionuclides: Part 1*, International Commission on Radiological Protection Publication No. 56, *Ann. ICRP* 20(2).

ICRP. 1991. *Recommendations of the International Commission on Radiological Protection* I; International Commission on Radiological Protection Publication No. 60. *Ann. ICRP* 21 (1–3)

ICRP. 1995. *Age-Dependent Doses to Members of the Public from Intake of Radionuclides, Part 5: Compilation of Ingestion and Inhalation Dose Coefficients*, International Commission on Radiological Protection Publication No. 72, *Ann. ICRP* 26(1). P 072 errata in SG 03 JAICRP 32 (1-2).

ICRP, 2006. *The Optimisation of Radiological Protection – Broadening the Process*, International Commission on Radiological Protection Publication 101b, *Ann. ICRP* 36 (3)6.

ICRP. 2009. *ICRP Statement on Radon and Lung Cancer Risk from Radon and Progeny*, Draft Report for Consultation, http://www.icrp.org/docs/ICRP_Statement_on_Radon_AND_Lung_cancer_risk_from_radon_and_progeny%28for_consultation%29.pdf.

Kocher D.C., Apostoaei A.I., Hoffman F.O. 2005. Radiation effectiveness factors for use in calculating probability of causation of radiogenic cancers. *Health Phys* 89(1): 3-32.

Linkov I., Burmistrov D. 2001. Reconstruction of doses from radionuclide inhalation for nuclear power plant worker using air concentration measurements and associated uncertainties. *Health Phys* 81(1): 70-75.

MSUE (Michigan State University Extension). 2011. *Agricultural Water Use Reporting.* Michigan State University Extension. Available at web1.msue.msu.edu/waterqual/WQWEB/Beef.doc.

National Research Council 2006. *Health Risks from Exposure to Low Levels of Ionizing Radiation, BEIR VII Phase II*, National Academies Press, Washington DC.

NCRP (National Council on Radiation Protection and Measurements). 1996. *A Guide for Uncertainty Analysis in Dose and Risk Assessments Related to Environmental Contamination*. National Council on Radiation Protection and Measurements Commentary No. 14, Bethesda, MD.

NCRP 1996, *A Guide for Uncertainty Analysis in Dose and Risk Assessments Related to Environmental Contamination*, NCRP Commentary No. 14, National Council on Radiation Protection and Measurements, Bethesda MD.

NCRP 1998, *Evaluating the Reliability of Biokinetic and Dosimetric Models and Parameters used to Assess Individual Doses for Risk Assessment Purposes*, NCRP Commentary No. 15, National Council on Radiation Protection and Measurements, Bethesda MD.

NCRP 2007, *Uncertainties in the Measurement and Dosimetry of External Radiation*, NCRP Report No. 158, National Council on Radiation Protection and Measurements, Bethesda MD.

NCRP 2009, *Uncertainties in Internal Radiation Dose Assessment*, NCRP Report No. 164, National Council on Radiation Protection and Measurements, Bethesda MD.

NCRP 2012, *Uncertainties in the Estimation of Radiation Risks and Probability of Disease Causation*, NCRP Report No. 171, National Council on Radiation Protection and Measurements, Bethesda MD.

NRC (United States Nuclear Regulatory Commission). 1993. *Final Environmental Impact Statement to Construct and Operate a Facility to Receive, Store, and Dispose of 11e.(2) Byproduct Material Near Clive, Utah*, NUREG 1476, US Nuclear Regulatory Commission, Washington, DC.

NRC. 2000. *A Performance Assessment Methodology for Low-Level Radioactive Waste Disposal Facilities*, NUREG 1573, US Nuclear Regulatory Commission, Washington, DC.

Puncher, M. and J.D. Harrison 2012, *Assessing the Reliability of Dose Coefficients for Ingestion and Inhalation of Radionuclides by Members of the Public*, Health Protection Agency, Center for Radiation, Chemical and Environmental Hazards, Chilton, Didcot, Oxfordshire OX11 0RQ, HPA-CRCE-048, ISBN 978-0-85951-741-6.

Puncher, M. and J.D. Harrison 2013, *Assessing the Reliability of Dose Coefficients for Inhaled and Ingested Radionuclides*. Journal of Radiological Protection 32:223-241.

Scott B.R. 2008. It's time for a new low-dose-radiation risk assessment paradigm- one that acknowledges hormesis. *Dose-Response* 6:333-351.

Staven L.H., Napier B.A., Rhoads K., Strenge DL. 2003. *A Compendium of Transfer Factors for Agricultural and Animal Products*, Pacific Northwest National Laboratory, Richland WA.

Tuli J. K., 2005. *Nuclear Wallet Cards, 7th Edition*, Brookhaven National Laboratory, Upton, NY.

UDWR (Utah Division of Wildlife Resources) 2009. *Utah Pronghorn Statewide Management Plan.*, Utah Department of Natural Resources, Salt Lake City UT.

USFS (United States Forest Service), 2005. *Off-Highway Vehicle Recreation in the United States, Regions, and States: A National Report from the National Survey on Recreation and the Environment.* June, 2005. US Forest Service, University of Georgia; Athens GA.

USFWS (United States Fish and Wildlife Service) 2006. *National Survey of Fishing, Hunting, and Wildlife-Associated Recreation: Utah*. US Fish and Wildlife Service, US Department of Commerce, and US Census Bureau; Washington, DC.

Utah 2010. *Standards for Protection Against Radiation*. Utah Administrative Code Rule R313 15. In effect on March 1, 2010. (http://www.rules.utah.gov/publicat/code/r313/r313-015.htm, accessed 17 Mar 2010).

# Appendix I: Discussion of Derivations of Selected Parameter Distributions

Distribution development utilized data where available, and exercised professional judgment where it was not available. For the parameter distributions discussed below, unless specified otherwise, the approach followed the *Probability Distribution Development* white paper (Appendix 14).

*Age*: Based upon the observed age quantile breakdown reported in USFS (2005) for recreational receptors, ignoring the age groups outside of the defined adult age range 16-60. For simplicity, and because age data specific to ranchers in the vicinity of Clive were unavailable, the same age distribution was also used for rancher receptors. The age range corresponds to bins used to aggregate ventilation rate data by EPA (2009b).

*Gender*: Based upon the observed percentage in USFS (2005).

*Body Weight*: EPA (2009a) reports body weights as quantiles, broken down by various age and gender categories. Mean body weight changes gradually with age, and is significantly different

between genders. A lognormal distribution was fit for each gender separately, with the log of the geometric mean was fit as a constant, a linear function of age, and a quadratic function of age, using the quantile likelihood fitting described in the *Probability Distribution Development* white paper (Appendix 14). The quadratic model produced the best fit, capturing the mean decrease in the population for the oldest age group:

$$\mu = \beta_0 + \beta_1 \text{Age} + \beta_2 \text{Age}^2 \tag{21}$$

where $e^{\mu}$ is the geometric mean.

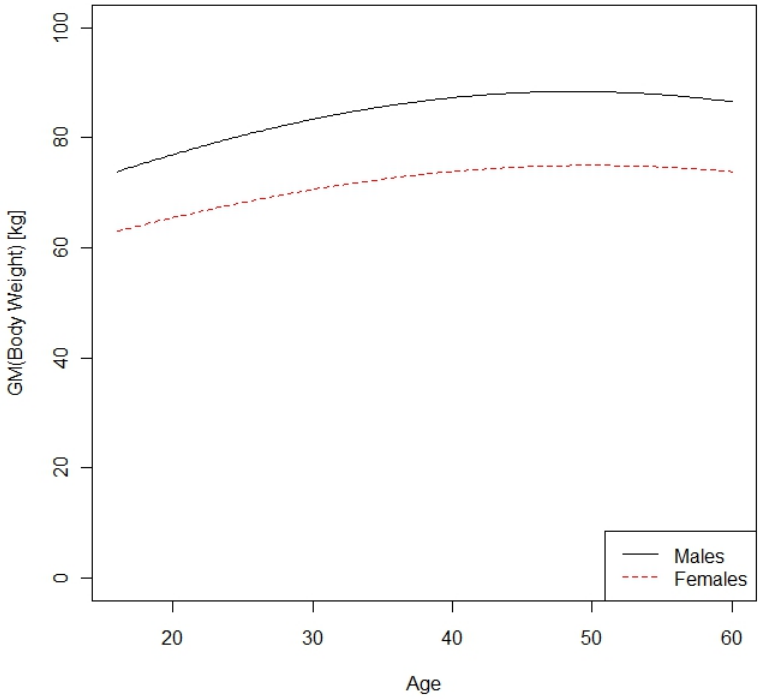


**Figure 1. Geometric mean of body weight as a function of age.**

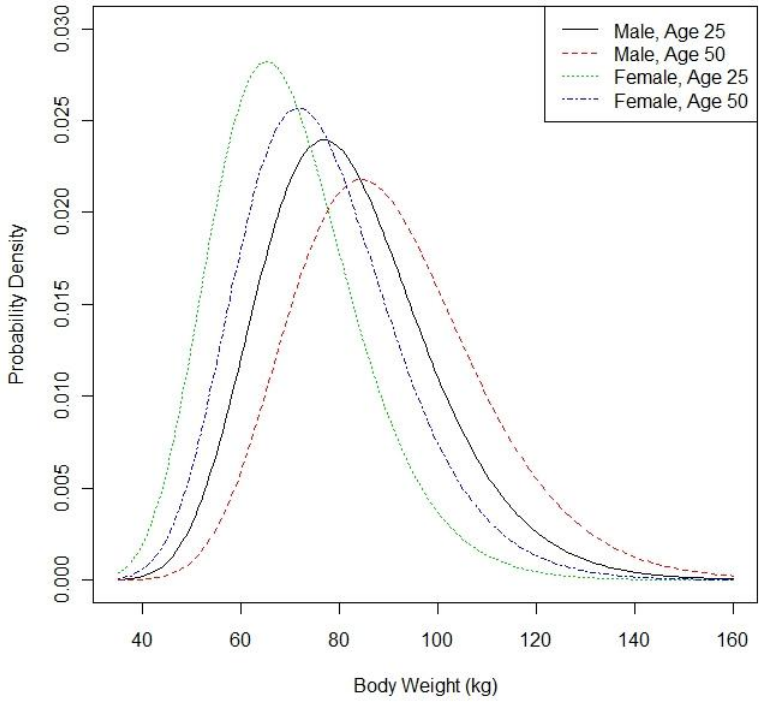


**Figure 2. Examples of distributions for body weight.**

*Ventilation Rate*: EPA (2009a) reports inhalation rates as quantiles, broken down by various activity, age, and gender categories. The data are reported as both weight-adjusted and non-weight-adjusted inhalation rates. In order to incorporate correlation in inhalation rates between activity categories, the weight-adjusted data are utilized. That is, a weight-adjusted inhalation rate will be simulated for each activity level, and then the single simulated body weight for the individual is multiplied by the weight-adjusted inhalation rates to obtain the inhalation rates:

$$V_{E,i} = V_{E,i}^{(BW)} \cdot BW \tag{22}$$

where $V_{E,i}$is inhalation rate for activity level $i$ in $m^3$/min, $V_{E,BW}$is body-weight adjusted inhalation rate for activity level $i$ in $m^3$/kg-min, and $BW$is body weight in kg. This approach to constructing inhalation rate is similar to the approach taken in EPA (2009b). Inhalation rate is significantly different between genders, and mean ventilation rate changes gradually with age. A lognormal distribution was fit for each gender separately. The log of the geometric mean was fit as a constant, a linear function of age, and a quadratic function of age; using the quantile likelihood fitting described in the *Probability Distribution Development for the Clive PA* white paper (Appendix 14). None of these models adequately characterized the data, as the 16-20 age group is significantly different from the 21-30 age group. As such, the 16-20 age group was fit separately from the remaining data, and a linear fit was adequate for the remaining age ranges.

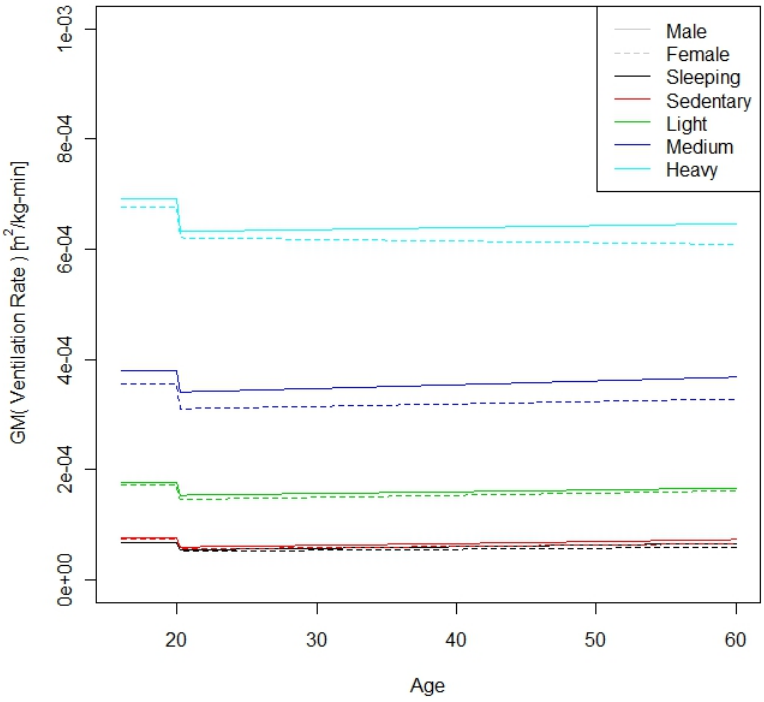


**Figure 3. Geometric means for ventilation rate, as a function of age and gender.**

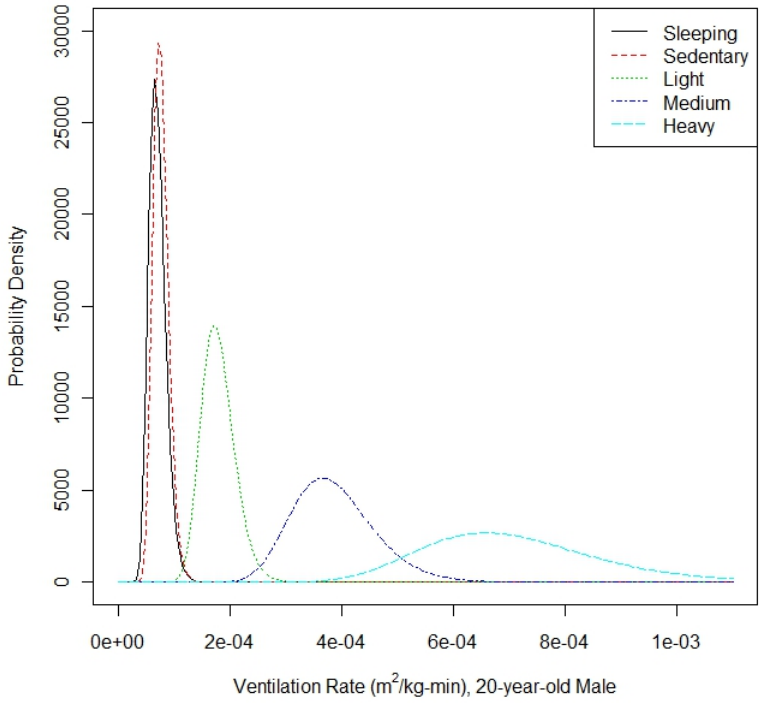


**Figure 4. Examples of ventilation rate distributions for different activities (20-year-old male).**

*Soil Ingestion Rate*: EPA (2009a) reports soil ingestion for adults only as a mean, median, and standard deviation. The distribution derived here is based upon the only careful study of adult ingestion that has been conducted to date (Davis and Mirick 2006), identified as a key study in EPA (2009a). Three tracer elements (aluminum, silicon, and titanium) used in Davis and Mirick (2006) provide different bases for quantifying soil ingestion rate. The data distribution is significantly different for the three tracer elements. Thus, rather than combine data across the three tracers, a separate distribution of soil ingestion is established for each tracer. Because there was no significant difference between genders, males and females were combined. Given the significant skew in the data (means much larger than the medians), a lognormal model was fit to the combined data based using maximum likelihood estimates.

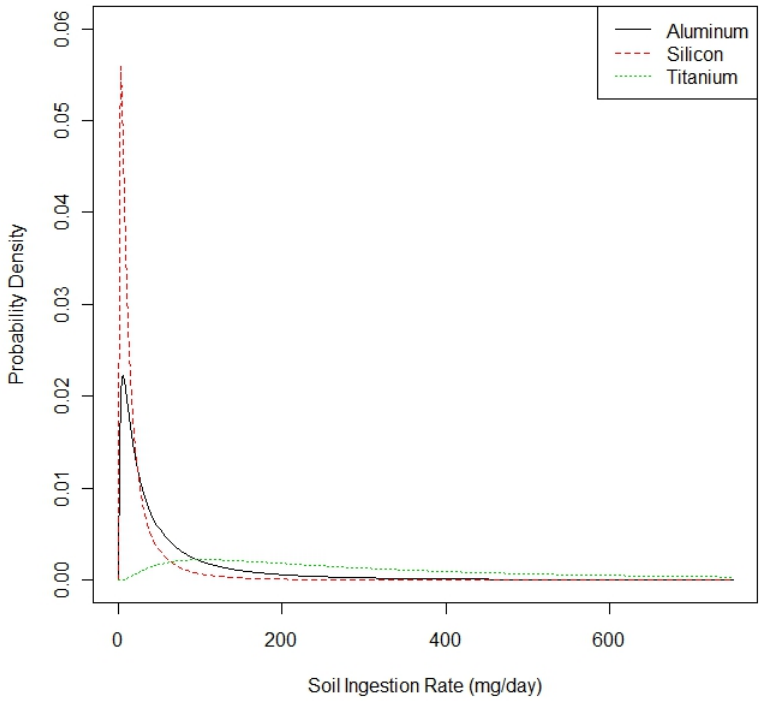


**Figure 5. Distributions for soil ingestion, representing different tracers.**

*Ingestion Rates, Home-produced Meat (beef)*: EPA (2009a) reports quantiles of the body-weight-adjusted average intake per day of home-produced meat, broken down by age and type of meat. The age groups given do not correspond perfectly to the range of ages considered in this PA model. Thus, the 20-39 age group was used to represent the 16-39 age group, and the 40-69 age group was used to represent the 40-60 age group. The distributions were significantly different for the two age groups, so they were fit separately. The lognormal distribution provided a good fit to the center of the data, but had poor tail behavior in each case. Thus, a gamma distribution was chosen instead, which provided a better overall fit.

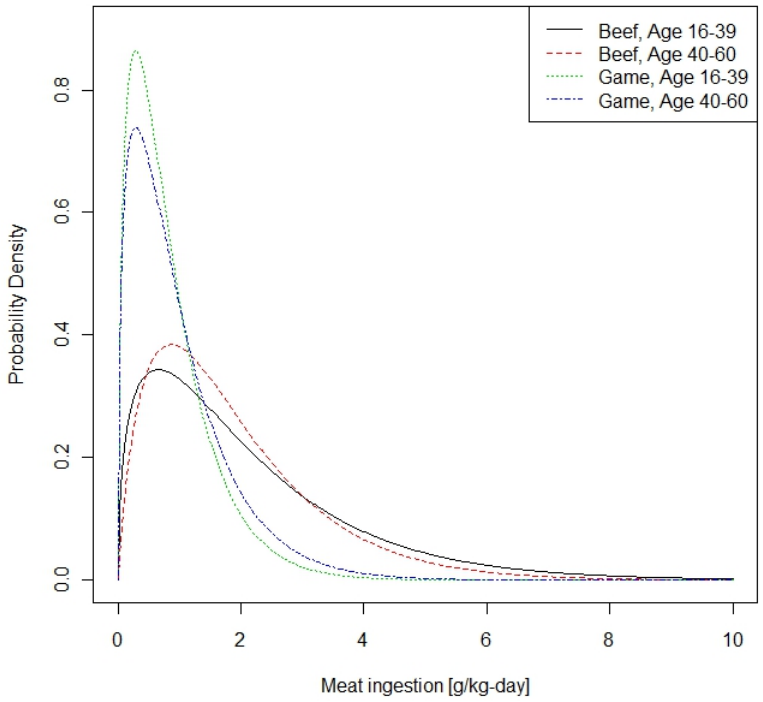


**Figure 6. Distributions for home-produced meat ingestion rates.**

*Activity-Based Exposure Time*: EPA (2009a) document reports average time per day spent at different levels of activity as quantiles for adults, broken down by age and gender. The quantiles are reported independently for each activity level, and thus no information regarding the correlation between the times is available. Correlation must exist, as an individual's daily averages must exist on the simplex that sums to 24 hours. Dirichlet distributions are the only standard statistical model that provides a distribution on a simplex. However, Dirichlet distributions could not achieve the long tails observed in the distributions for the more active levels. In order to achieve the tail behavior, the following approach was used. A lognormal model was fitted for combined sleeping and sedentary time (constrained to be no more than 24 hours). Sleeping time alone was also fitted as a lognormal model and constrained to be smaller than combined sleeping and sedentary time. Remaining average time per day was then partitioned into light, medium, and heavy activities. A lognormal distribution was fit to each, but for simulation purposes, the three values are simulated and then normalized to sum to time per day remaining. The resulting distribution induces moderate negative correlation amongst the time spent in each activity level; the greatest negative correlation existing between light and medium activity durations. The tail behavior of medium and heavy activity durations is reduced from that observed in the data (i.e., the upper percentiles are slightly lower than observed). However,

without the detailed correlation structure of the data, a simple model is unlikely to both meet the constraints of the simplex and match the tail behavior.

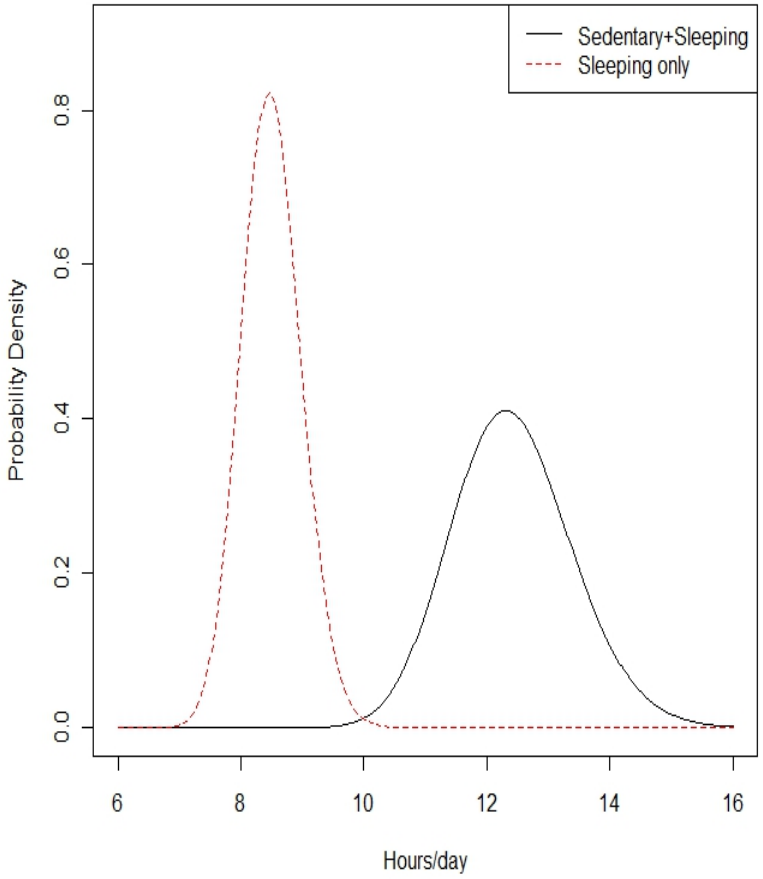


**Figure 7. Example distributions for sedentary plus sleeping time/day and sleeping time/day (30-year-old female).**

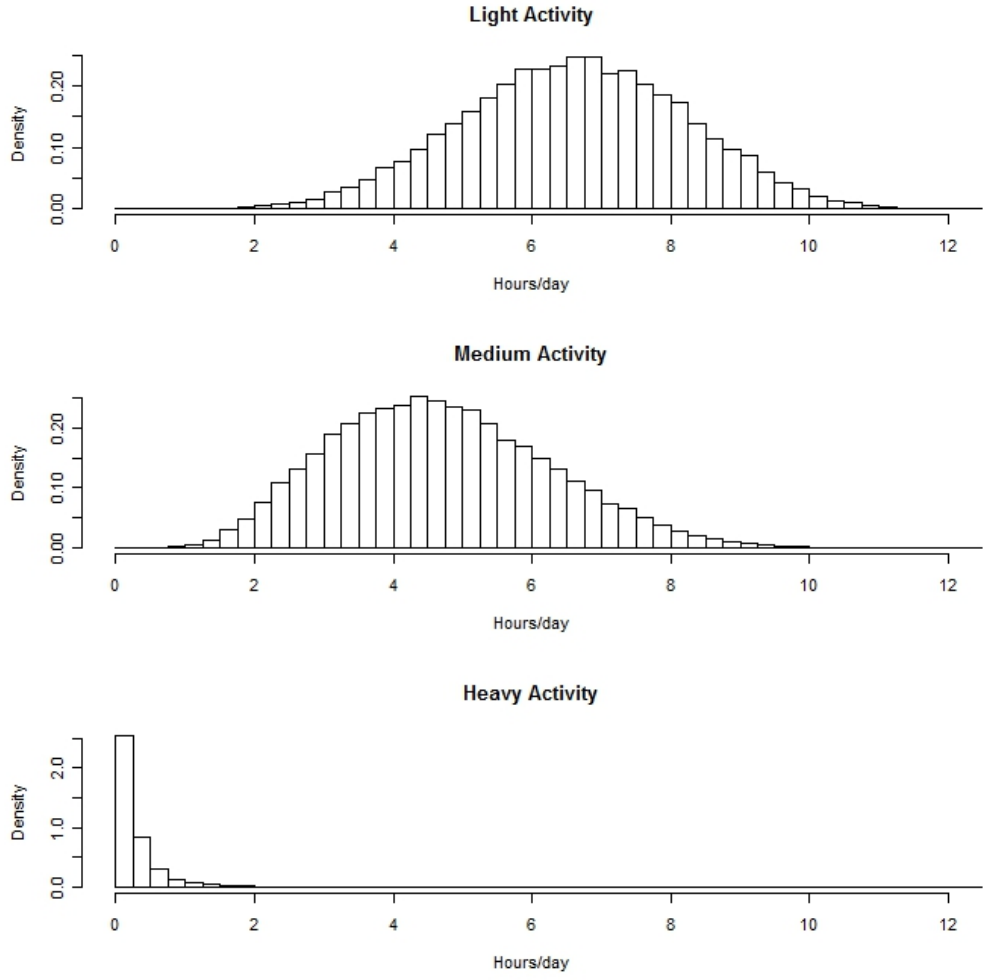


**Figure 8. Distributions for light, medium, and heavy activity time/day (30-year-old female).**

*Numbers of Individuals in Vicinity of Site* – Personal communication with BLM staff (Salt Lake Field Office) provided 100 and 500 as bounds and 350 as a best guess. These might be interpreted as 5$^{th}$ and 95$^{th}$ percentiles, along with a mean or median. However, due to the informal nature of the conversation and a programming need to have a fixed upper bound on this distribution, these will be treated as bounds, making a triangular distribution a reasonable representation of the information.

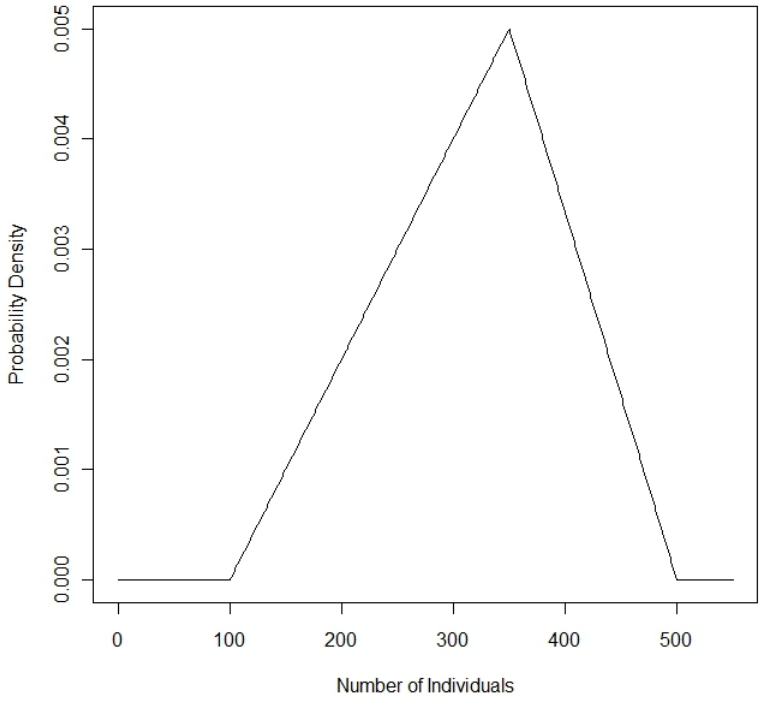


**Figure 9. Distribution for the total number of individuals at the site during a given year.**

*Receptor Type* – The individuals in the vicinity of the site are partitioned into Ranchers, Hunters, and Sport OHVers. The distribution for the number of Ranchers was based upon professional judgment and the size of leases, and is independent of the total number of individuals within vicinity of the site. The remaining individuals are then partitioned into Hunters and Sport OHVers by utilizing a binomial distribution with the proportion of hunters equal to 0.25, the value reported from the large survey in USFS (2005).

*Sport OHVer Day-Trip Time in Area* – The only reported value from the Sport OHVer survey was a mean of 6.3 hr/day. The standard deviation is not reported, so professional judgment was used to choose a standard deviation.

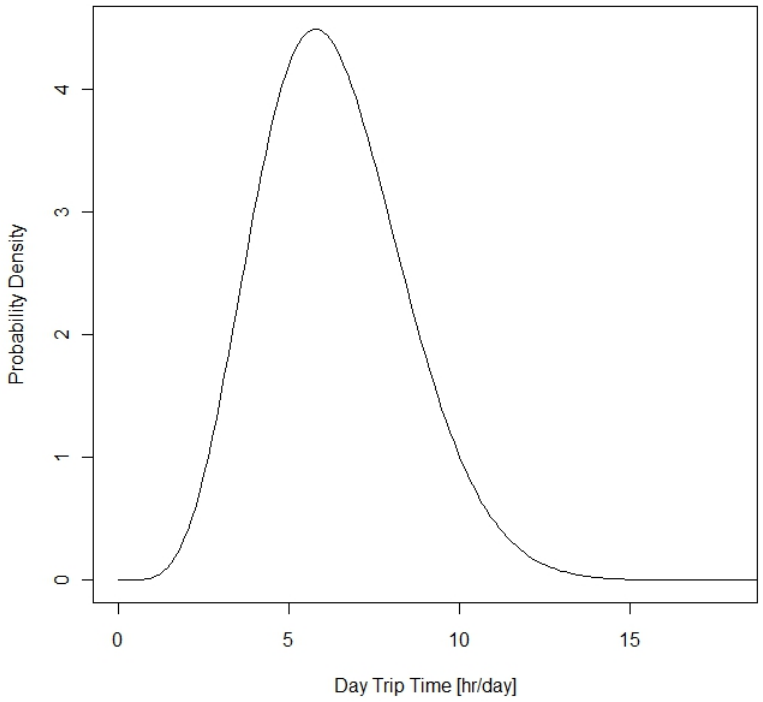


**Figure 10. Distribution for the average day-trip time.**

*OHV Dust Loading* – Summary data from EPA (2008) are available both for ambient conditions (CCMA) and near ATV riders. Means are given, and standard errors for the mean can be approximated from the upper confidence limit (UCL) values, by assuming a *t*-UCL. The standard errors are high relative to the mean, so each of these distributions was treated as lognormal. These two distributions were then simulated and a ratio taken, to obtain a distribution on the ratio. The resulting distribution is also approximately lognormal. Figure 11 shows the simulated values, along with the fitted distribution.

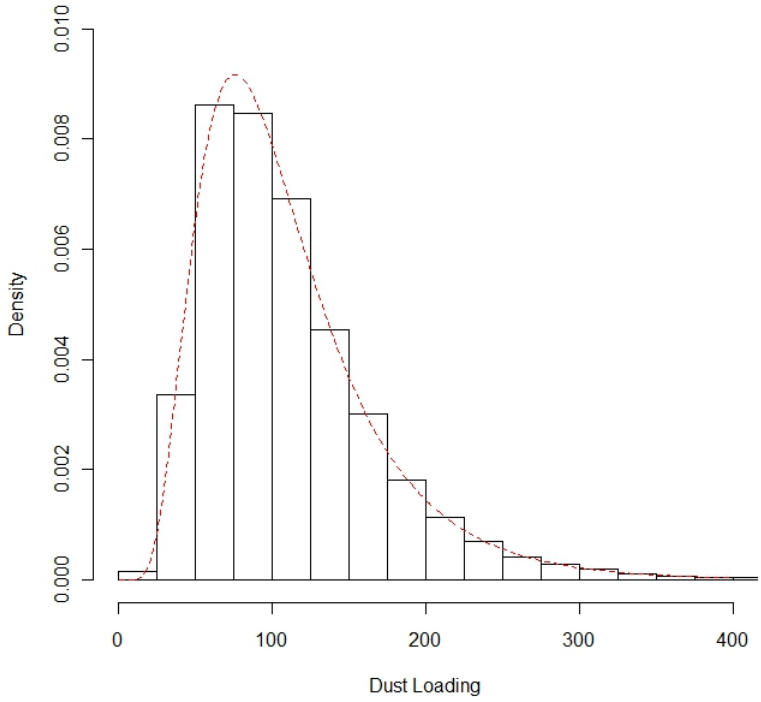


**Figure 11. Distribution for dust loading (overlaid on a histogram of simulated values).**

*Rancher Exposure Frequency* – Grazing leases are granted for 180 days each year, giving a natural upper bound for the distribution. There is little other information available to develop a distribution, so professional judgment was used, and a distribution was chosen that has most Ranchers spending a high proportion of the allotted 180 days on site, but allows for Ranchers that spend weekends off-site, do not utilize their full lease, etc.

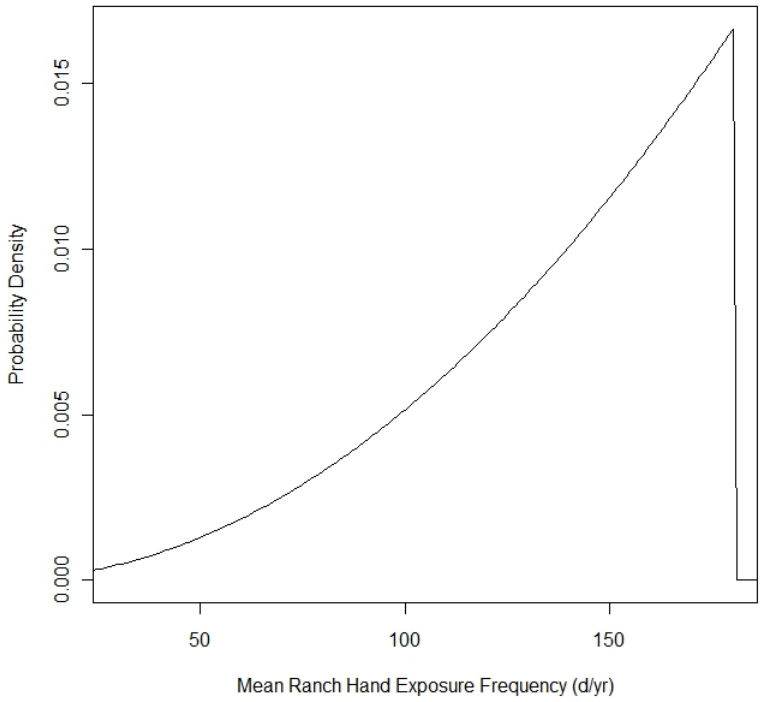


**Figure 12. Distribution for Rancher exposure frequency.**

*Sport OHVer Exposure Frequency* – The USFS (2005) document reports a confidence interval for the mean exposure frequency, which can be used to calculate the standard deviation of the exposure frequency. Because the standard deviation is larger than the mean, a lognormal model was used to match the observed mean and standard deviation from the survey data.

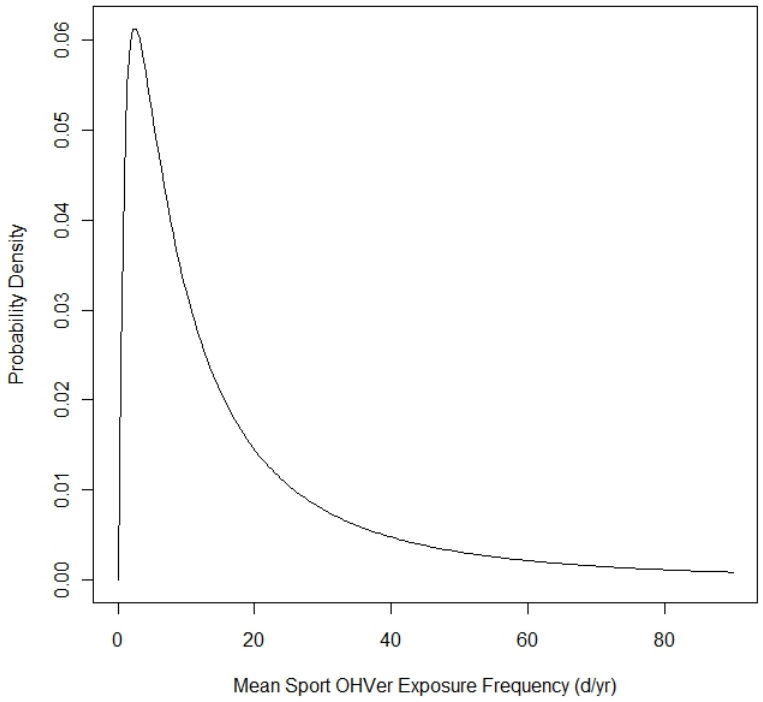


**Figure 13. Distribution for Sport OHVer exposure frequency.**

*Hunter Exposure Frequency* – The USFWS (2006) provides a mean estimate of 10 d/yr, but does not provide any other summary information. It may be reasonable to assume that this distribution has a similar shape as the exposure frequency for Sport OHVers; i.e., a right-skewed distribution that has most of the population spending a relatively small amount of time, with a few individuals who dedicate a great deal of time to the activity. Thus, a lognormal distribution was chosen with a mean of 10 d/yr, and a geometric standard deviation that matches the Sport OHVer geometric standard deviation.

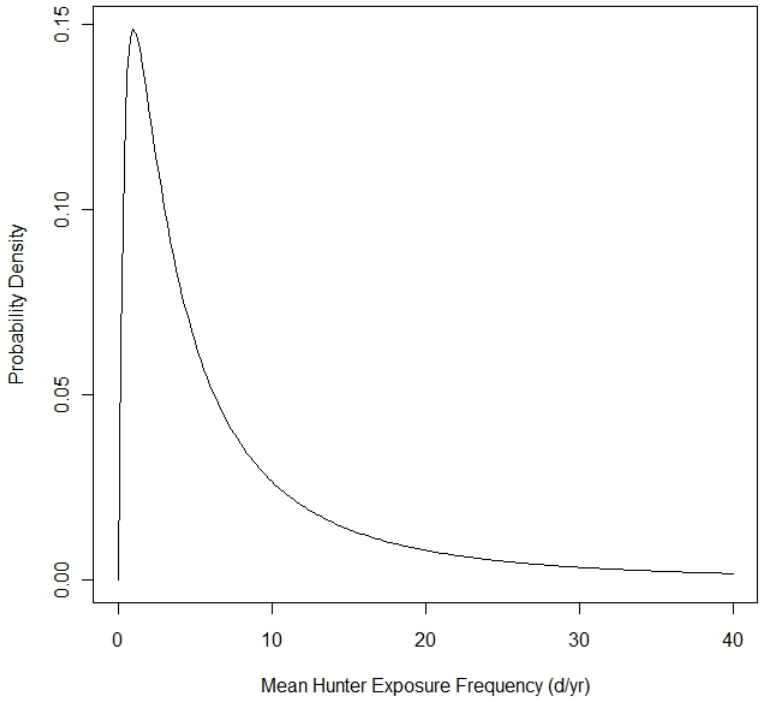


**Figure 14. Distribution for Hunter exposure frequency.**

*Rest Area Caretaker Exposure Frequency* – The distribution for this parameter was based on professional judgment. The maximum was conservatively set to the maximum possible exposure of 365 days per year. The mode is set to the EPA default exposure value of 350 days per year, and the minimum allows for 28 days of vacation plus 10 holidays for which the caretaker would be off-site.

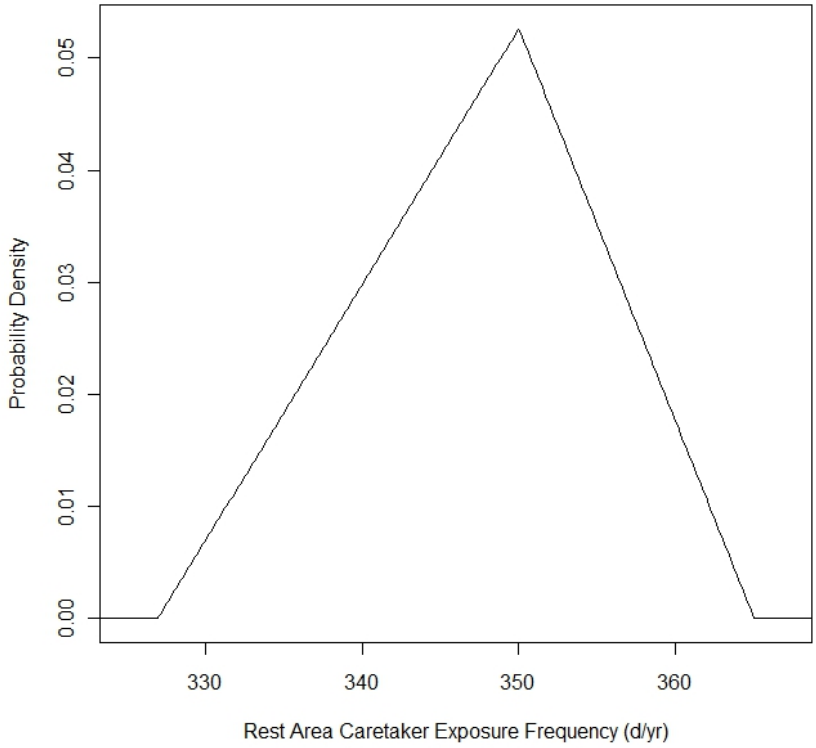


**Figure 15. Distribution for rest area caretaker exposure frequency.**

*Meat Loss* – EPA (1997b) provides information on the amount of meat lost in preparation and in post-cooking. An average and a standard deviation are reported for the *mean* loss. As the distribution of interest represents uncertainty about he mean, the average and standard deviation were used for a normal distribution.

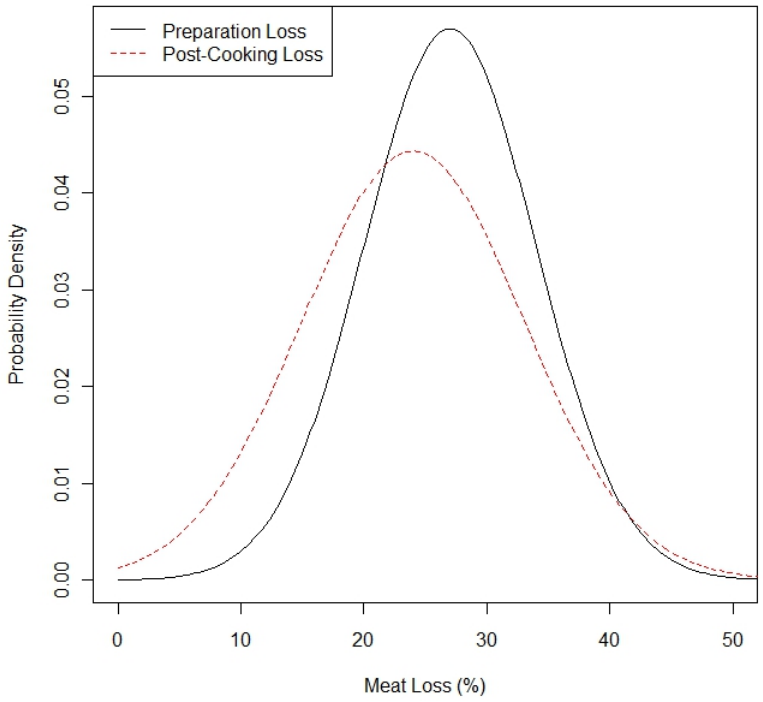


**Figure 16. Distributions for meat loss (preparation and post-cooking).**

*Cattle Range Acreage* – There are only four data points available (the four leases in the Clive area), but because the distribution of the mean acreage is desired, the mean and standard error of the mean are used to define a normal distribution.

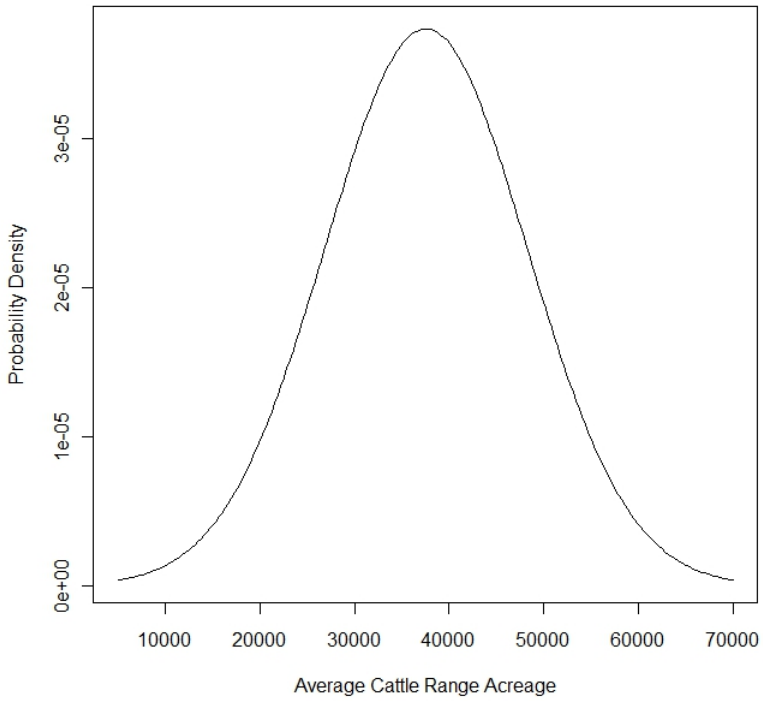


**Figure 17. Distribution for the average cattle range acreage.**

*Miscellaneous Uniform Distributions* – For many of the parameters, little information is available that is specific to the Clive facility site. A default distribution in such a case was a uniform distribution over a range of theoretical values, or from the minimum and maximum values found in literature. The uniform distribution is generally a poor representation of uncertainty but has the advantage of spreading its mass across a range of possible values. These uniform distributions are used as defaults until a sensitivity analysis can be performed to demonstrate whether further data collection is needed to construct a better representation of uncertainty.

*REF Distributions* - Kocher (2005) utilized lognormal distributions to represent the uncertainty in REF parameters. Thus, lognormal distributions were fit to the reported 2.5th, 50th, and 97.5th percentiles of these distributions.

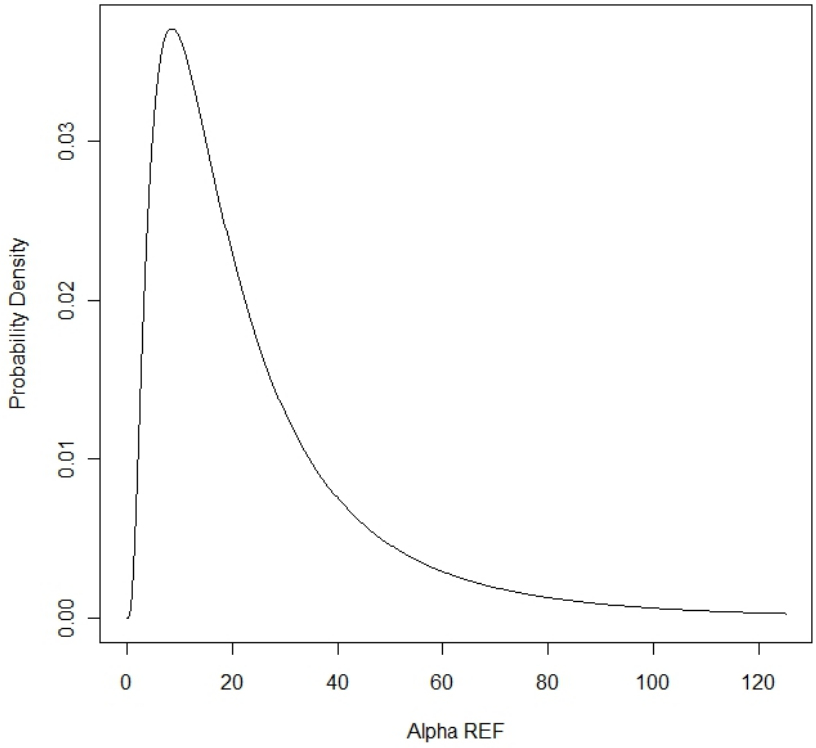


**Figure 18. Distribution for alpha particle REF.**

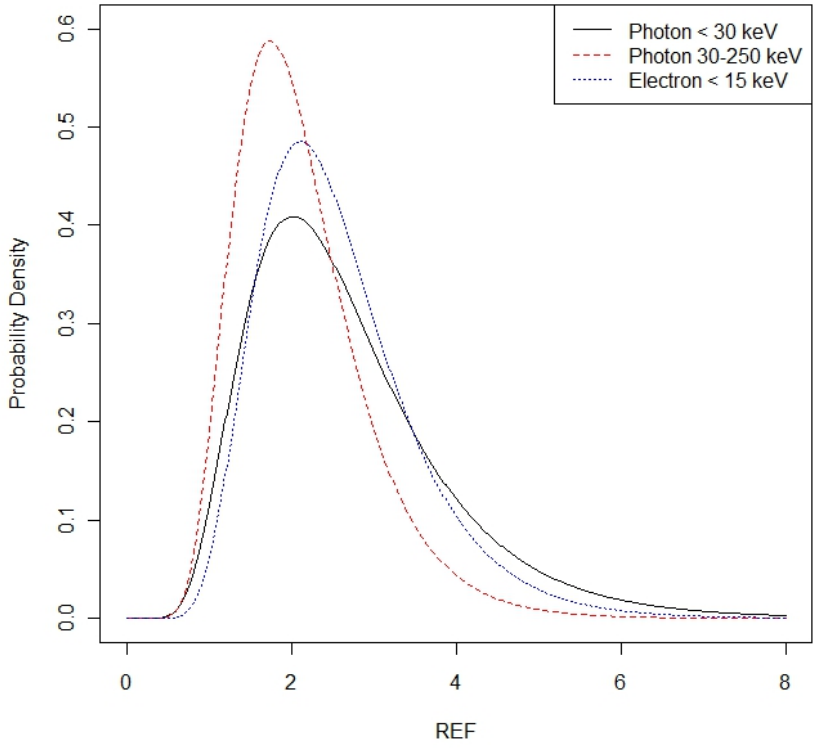


**Figure 19. Distribution for electron and photon REFs.**

*Uranium oral reference dose* – EPA has two published values for this value: EPA (2011) and EPA (2000). These two sources are considered equally viable, so each is selected with 50% probability.