# Final Report for the Clive DU PA Model version 1.0

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Prepared by

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# CONTENTS

# FIGURES

# TABLES

# Executive Summary

Neptune and Company, Inc., under contract to Energy*Solutions*, LLC (Energy*Solutions*), has developed a computer model (the Clive DU PA Model—the Model) to support decision making related to the proposed disposal of depleted uranium (DU) wastes at the low-level radioactive waste (LLW) disposal facility at Clive, Utah, operated by Energy*Solutions*. The model provides a platform on which to conduct analyses relevant to performance assessment (PA), as required by the State of Utah in Utah Administrative Code (UAC) R313-25, License Requirements for Land Disposal of Radioactive Waste (Utah 2010). Specifically, a PA is required in UAC R313-25-8, Technical Analyses. The model may also serve to inform decisions made by the Site operator in order to gain maximum utility of the resource that is the Clive Facility.

Depleted uranium is the remains of the uranium enrichment process, of which the fissionable uranium isotope $^{235}U$ is the product. The leftover uranium, depleted in $^{235}U$, is predominantly $^{238}U$, but may include small amounts of other U isotopes. In general, DU will contain very small amounts of decay products in the uranium, thorium, actinium, and neptunium series of decay chains. Some specific DU waste, resulting from introduction of uranium retrieved from used nuclear reactor fuel (reactor returns) into the separations process, contains varying amounts of contaminants, in the form of fission and activation products. Since the DU is not all pure uranium and its decay products, it is here termed "DU waste". The national inventory of DU is on the order of 700 Gg (700,000 Mg, or metric tons) in mass, and the bulk of it exists in its original storage cylinders as uranium hexafluoride ($DUF_6$), awaiting conversion to an oxide form ($U_3O_8$) for disposal. This conversion is to be performed at the Portsmouth, Ohio, and Paducah, Kentucky gaseous diffusion plant (GDP) sites, using new purpose-build "deconversion" plants. A much smaller mass of DU waste was generated by the Savannah River Site (SRS) in the form of $UO_3$, a powder stored in several thousand 200-L (55-gal) drums. While the composition of the SRS DU is reasonable well known, the content of the GDP DU is not well documented. For the purposes of this assessment, it was necessary to assume that some uncertain fraction of the GDP DU waste was contaminated to the same extent as the SRS DU. DU waste from both sources is considered in the Clive DU PA Model.

The Model is written using the GoldSim probabilistic systems analysis software, which is well-suited for the purpose. In order to provide decision makers with a broad perspective of the behavior and capabilities of the Facility, the model considers uncertainty in input parameters and to some extent in modeling approaches. This probabilistic assessment methodology is encouraged by the Nuclear Regulatory Commission (NRC) and the Department of Energy (DOE) in constructing PAs and the models that support them. The Model can be run in deterministic mode, where a single set of model inputs is used, but running in probabilistic *Monte Carlo* mode provides greater insight into the model behavior, and especially into model sensitivity. In *Monte Carlo* mode, a large number of equally-probably realizations are executed, and the results reflect the uncertainty in the model. To the extent that the model reflects the uncertain state of knowledge at a site, the model provides insight about how the site works, and what should be expected if different actions are taken or different wastes are disposed. In this way, the model aids in decision making, even in the face of uncertainty.

The Clive Facility is located at the eastern edge of the Great Salt Desert, west of the Cedar Mountains, and approximately 100 km (60 mi) west of Salt Lake City, Utah. Clive is a remote and environmentally inhospitable area. Human activity at Clive has, historically, been very limited, due largely to the lack of potable water, or even water suitable for irrigation. The site is located on flat ground, with the bottom of the waste disposal cells shallowly excavated into local lacustrine silts, sands, and clays. A single waste disposal cell, or embankment, is considered in this model: the Class A South embankment. This is modeled with an engineered cover, as per design documents. The top of the cell is above grade, and the cover has layers of engineered materials of earthen origin. In time, this cover is expected to become infilled with loess (windblown silt from local lacustrine deposits), vegetated with native plants, and occupied to a limited extent by insects and mammals. As plant communities become established, they are likely to keep the cover system fairly dry through transpiration.

Some water is modeled as penetrating the cover system, however, and this infiltration leaches radionuclides and transports them down through the cell liner and unsaturated zone to the aquifer. In the saturated zone (aquifer), contaminants are transported laterally to a hypothetical monitoring well located about 27 m (90 ft) from the edge of the interior of the cell. Since the side slopes of the cell are modeled to not contain DU waste, the effective distance to the well from the DU waste itself is about 73 m (240 ft). This pathway is significant for long-lived and readily-leached radionuclides such as $^{99}$Tc. Contributions to groundwater radionuclide concentration from the proposed DU waste are calculated for comparison to groundwater protection limits (GWPLs) during the next 500 years (UWQB 2009).

In addition to water advective transport, radionuclides are transported via diffusion in both water and air phases, which can provide upward pathways. Gaseous radionuclides, such as $^{222}$Rn, partition between air and water. Soluble constituents partition between water and solid porous media. Coupled with all these process are the activities of biota, with plants transporting contaminants to the ground surface in their tissues, and burrowing animals (ants and small mammals) moving bulk materials upward and downward through burrow excavation and collapse. Biota do not play a major role in contaminant transport, according to model results, but the potential effect of black greasewood, with its long tap root, is occasionally apparent. The cover, with its upper layers infilled with loess, will be largely self-healing from the effects of roots, burrows, and desiccation, but the degree to which the compacted clay radon barriers at the bottom of the cover would be affected is not well understood. The model does not consider the effects of enhanced infiltration or radon diffusion from a compromised radon barrier.

Once radionuclides reach the ground surface at the top of the engineered cover, they are subject to suspension into the atmosphere and dispersion to the surrounding landscape. Atmospheric transport of gases ($^{222}$Rn) and contaminants sorbed to suspended particles is modeled using a standard modeling platform approved by the Environmental Protection Agency (EPA), called AERMOD. The results of this model are abstracted into the Clive DU PA Model, and contributions of airborne radionuclides to dose and uranium toxicity hazard are evaluated.

The potentially significant cover degradation process of gully formation is evaluated using a simple modeling construct, in order to determine whether it warrants more sophisticated modeling approaches. The Model employs the geometry of gully formation. It is assumed that a gully could form, as the result of natural or anthropogenic processes, as a wedge-shaped incision into the cover, with the top end at the cover ridge, and the mouth at the change in slope. Outwash from the gully forms a fan-shaped deposit on the side of the embankment. A small number of gullies (1 to 20) is posited, to determine if the number of gullies is significant. Materials exposed in the gully bottom are presumed to be spread across the top of the fan. If these materials include DU waste components, then this leads to some contribution to doses and uranium hazards. No associated effects, such as biotic processes, effects on radon dispersion, or local changes in infiltration are considered. When gullies encounter DU waste, doses and uranium hazards are increased, but when wastes are buried sufficiently deep the gullies have essentially no effect on human exposures.

Given the remote and inhospitable environment of Clive, it is not reasonable to assume that the traditional residential receptors considered in PA will be present. Traditionally, and based on DOE (DOE M 435.1) and NRC guidance (10 CFR 61), members of the public are evaluated outside the fence line or boundary of the disposal facility, and inadvertent intruders are assumed to access the disposal facility and the disposed waste directly, in activities such as well drilling or house construction. For disposal facilities in the arid west, these types of strictly defined default scenarios do not adequately describe likely human activities. Their inclusion in a PA for a site in the arid west, such as Clive, will usually result in underestimation of the performance of a disposal system, which does not lend itself to effective decision making for the Nation's needs to dispose of radioactive waste.

At Clive, there is no potable water resource to drill for, and historical evidence suggests there is little likelihood that anyone would construct a residence on or near the site. There are present day activities in the vicinity, however, that might result in receptor exposures if these activities are projected into the future when the facility is closed and after institutional control is lost. Large ranches operate in the area, so ranch hands will work in the vicinity. Pronghorn antelope are found in the region, and hunters will follow them. Both of these activities are facilitated by the use of off-highway vehicles (OHVs). OHV enthusiasts also ride recreationally for sport in areas adjacent to the facility.

In addition to these receptors, there are specific points of exposure within the vicinity of the Clive Facility where individuals might be exposed. About 12 km (8 miles) to the west, OHV enthusiasts use the Knolls Recreation Area. Interstate-80 and a railroad are located to the north, with an associated rest area on the highway. Closer to the Clive Facility, the Utah Test and Training Range access road is used on occasion. The Model hence evaluates dose, or risk, to site-specific receptors.

The State of Utah follows federal guidance by categorizing receptors in a PA in UAC Rule R313-25-8 and 10 CFR 61.41 according to the labels "member of the public" (MOP) and "inadvertent human intruder" (IHI). NRC offers two definitions of inadvertent intruders in 10 CFR 61:

> **§ 61.2 Definitions**. *Inadvertent intruder* means a person who might occupy the disposal site after closure and engage in normal activities, such as agriculture, dwelling construction, or other pursuits in which the person might be unknowingly exposed to radiation from the waste.

> **§ 61.42 Protection of individuals from inadvertent intrusion.** Design, operation, and closure of the land disposal facility must ensure protection of any individual inadvertently intruding into the disposal site and occupying the site or contacting the waste at any time after active institutional controls over the disposal site are removed.

NRC offers one reference to an MOP:

> **§ 61.41 Protection of the general population from releases of radioactivity.** Concentrations of radioactive material which may be released to the general environment in ground water, surface water, air, soil, plants, or animals must not result in an annual dose exceeding an equivalent of 25 millirems [0.25 mSv] to the whole body, 75 millirems [0.75 mSv] to the thyroid, and 25 millirems [0.25 mSv] to any other organ of any member of the public. Reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable.

DOE definitions in DOE M 435.1 (the Manual accompanying DOE Order 435.1) are much more specific. However, the applicable federal agency that regulates disposal of low-level radioactive waste at the Clive Facility is NRC. For the Clive Facility and the Model, based on the NRC definitions, the ranch hand, hunter and OHV enthusiast are expected to engage in activities both on and off the site. As such, these receptors fit the NRC definition of inadvertent intrusion. This is the case whether or not gullies are included in the model, although inclusion of gullies presents a mechanism for more direct intrusion into the DU waste. The receptors that are located at specific offsite locations, instead, fit the NRC definition of member of the public. The Model presents predicted doses to the receptors identified above, under the conditions and assumptions that provide the basis for the Model. These doses are presented as the results of the Model. The effect of comparison with MOP and IHI performance objectives is also presented.

The Model addresses radiation dose to human receptors who might come in contact with radionuclides released from the disposal facility into the environment subsequent to facility closure. In accordance with UAC Rule R313-25-8, doses are calculated within a 10,000-year compliance period and may be compared to a performance criterion of 25 mrem in a year for a MOP, and 500 mrem in a year for an inadvertent intruder. The dose assessment component of the PA model, like the transport modeling components described above, supports probabilistic *Monte Carlo* analysis. Spatio-temporal scaling is a critical component of the Model development. For example, the Model differentiates the impact of short-term variability in exposure parameters (values applicable over a few years or decades, such as individual physiological and behavioral parameters) from the longer-term variability of transport parameters (values applied over the full 10,000-year performance period, such as hydraulic and geochemical parameters). This distinction facilitates assessment of uncertainties that relate to physical processes from uncertainties relating to inter-individual differences in potential future receptors.

In addition to radiation dose, uranium is also associated with non-radiological toxicity, e.g. kidney damage. The potential chemical toxicity of uranium disposed at the Clive Facility is evaluated in the Model. Potential receptor exposure to uranium is compared to toxicological criteria that pertain to a threshold of adverse effect associated with kidney toxicity.

These doses and the supporting contaminant transport modeling that provides the dose model with radionuclide concentrations in exposure media, are evaluated for 10,000 yr, in accordance with UAC R313-25-8(2). After that time, active contaminant transport and exposure modeling is no longer useful, and the focus turns to long-term, or “deep time” scenarios. Peak activity of the waste occurs when the principal parent $^{238}U$ (with a half-life that is approximately the age of the earth— over 4 billion years), reaches secular equilibrium with its decay products. This occurs at roughly 2.1 My from the time of isotopic separation, and the model evaluates the potential future of the site in this context. This time frame borders on geologic, and needs to take into account the likely possibility of future large lakes in the Bonneville Basin. The return of such lakes is understood to be inevitable, and the Clive Facility, as constructed, will not survive in its current configuration. Many lakes, of intermediate and large size, are expected to occur in the 2.1-My time frame, following the climate cycle periodicity of about 100,000 yr, based on current scientific understanding of paleoclimatology.

As each lake returns, estimates are made of the radionuclide concentrations in a local part of the lake, and in the sediments surrounding and subsuming the site. Because the exact behavior of lake intrusion and site destruction is speculative, the model makes several conservative assumptions. The entirety of the DU waste is assumed to comingle with sediments, dispersed over an uncertain area. In the presence of a lake, the radionuclides migrate into the water column, in accordance with their aqueous solubility. For $U_3O_8$, which is considered to be the only form of uranium oxide remaining by the time the first lake arrives (since $UO_3$ has a relative high solubility and will be washed out of the embankment in roughly 50,000 yr), the solubility of U is very low, so its sediment concentration is relatively high. As each lake recedes, radionuclides are co-deposited with the sediment, only to be dissolved into the water column again with the next lake. This is a very conservative approach, since in reality each blanket of sediment could entrap constituents, and the concentrations in water and sediment over time should decrease consequently. The analysis, therefore, focuses on the arrival of the first lake, which will be the most destructive in terms of sudden release of radionuclides, and would provide the least amount of sediment to encapsulate them. Subsequent lakes would see progressively less radionuclide activity as the site is slowly buried under ever-deeper lacustrine deposits through the eons.

The utility of such a calculation, aside from responding to the UAC, is to inform decisions regarding the placement of wastes in the embankment. With downward pathways influencing groundwater concentrations, and upward pathways influencing dose and uranium hazard, a balance must be achieved in the placement of different kinds of waste. The Model reported herein includes three different options for configuration of the DU waste within the CAS embankment. The volume within the embankment that is available for waste disposal is about 13.5m deep below the engineered cap. The 13.5m is divided into 27 layers that are all 0.5m thick. The layers are labeled 1 through 27 from top to bottom of the available volume. No DU waste is included under the side slopes for this PA.

1. GDP contaminated waste in Layer 7 – SRS waste in Layer 8 – GDP uncontaminated waste in Layers 9-27. This model is termed the 3-m model, because the top of Layer 7 is 3 m below the embankment cover. Note that clean fill material is assumed for the 3 m between the cap and Layer 7.

2. GDP contaminated waste in Layer 11 – SRS waste in Layer 12 – GDP uncontaminated waste in Layers 13-27. This model is termed the 5-m model, because the top of Layer 11 is 5 m below the cap. Note that fill material is assumed for the 5 m between the cap and Layer 11.

3. GDP contaminated waste in Layer 21 – SRS waste in Layer 22 – GDP uncontaminated waste in Layers 23-27. This model is termed the 10-m model, because the top of Layer 21 is 10 m below the cap. Note that fill material is assumed for the 10 m between the cap and Layer 21. This model places all waste below grade.

These options cover a fairly wide range of possible disposal options, from disposal below grade only to disposal throughout most of the system, which helps explore the range of possible options for disposal of DU waste. In addition to these options, two scenarios are considered that are related to erosion. The first essentially assumes a stable embankment for 10 ky, with infilling of the cap and continual airborne deposition replacing fine sediments that are resuspended themselves and subsequently dispersed offsite. This model assumes a balance so that substantial erosion from air and water borne forces is unlikely. The second scenario is one in which gullies are formed that, depending on the DU waste disposal configuration, might intersect and expose the DU waste to the environment. Consequently, six different models are considered for the dose and groundwater concentration endpoints. Dose results for ranch workers are presented in Tables ES-1 (without gullies) and ES-2 (with gullies). Doses to ranch workers are more than an order of magnitude greater than doses to hunters and OHV enthusiasts. Groundwater results for $^{99}$Tc in Table ES-3.

There is a question of which statistic is most appropriate for comparison. The statistics in Tables ES-1 and ES-2 represent summaries of the peak of the mean doses. If the model is constructed properly, and considering that doses increase with time given the model construction and assumptions so that the peak mean dose occurs at or near 10 ky, then the 95$^{th}$ percentile is analogous to the 95% upper confidence interval of the mean that is commonly used to represent reasonable maximum exposure in CERLCA risk assessments. The mean, instead represents a central tendency estimate of risk under CERCLA.

When gullies are not included in the model, compliance with the performance objectives for the inadvertent intruder of 500 mrem in a year, and for the MOP of 25 mrem in a year is clearly established for all three disposal configurations. The doses increase as waste is placed nearer the top of the embankment, but the more stringent MOP performance objectives are not exceeded in all cases. This implies that disposal configurations exist, under the conditions of this model, for which it is reasonable to dispose of DU waste.

When gullies are included (Table ES-2), all doses are still less than the 500-mrem in a year inadvertent intruder performance objective. However, the 95$^{th}$ percentile peak mean dose to ranch workers exceeds the MOP performance objective of 25 mrem in a year.

Results are also available for the offsite (MOP) receptors. None of the 95$^{th}$ percentile dose estimates for these receptors exceeds 1 mrem in a year, and most of the peak mean dose estimates are much less than 1 mrem in a year.

**Table ES-1. Peak mean TEDE, without consideration of gullies: statistical summary**

| | Peak TEDE (mrem in a yr) within 10,000 yr* | | |
|---|---|---|---|
| **receptor** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 4.37 | 3.44 | 11.3 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.598 | 0.473 | 1.52 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00596 | 0.00471 | 0.0152 |

* - Results based on 5,000 simulations of the Model

**Table ES-2. Peak mean TEDE, with gully screening calculation: statistical summary**

| | Peak TEDE (mrem in a yr) within 10,000 yr* | | |
|---|---|---|---|
| **receptor** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 20.9 | 11.6 | 72.3 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.564 | 0.443 | 1.44 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00594 | 0.00457 | 0.0155 |

* - Results based on 5,000 simulations of the Model

Summary statistics for the distribution of the peak of the mean $^{99}$Tc concentrations are presented in Table ES-3. For the 3-m and 5-m models, compliance with the GWPLs is clearly demonstrated. For the 10-m model the situation is not as clear. However, both the mean (of the peak of the means) and the $95^{th}$ percentile exceed the GWPL, in which case, it is probably not unreasonable to conclude that the 10-m model is not in compliance with the performance objective.

The results depend critically on the model structure, specification and underlying assumptions. Infiltration rates and $^{99}$Tc inventory concentrations might be overestimated. However, based on the model assumptions the 10-m model does not comply with the GWPL performance objective for $^{99}$Tc. These results suggest, however, that there are configurations that comply with the GWPLs.

**Table ES-3. Peak groundwater activity concentrations for 99Tc within 500 yr, compared to GWPLs**

| radionuclide | GWPL (pCi/L) | peak activity concentration within 500 yr (pCi/L)* | | |
|---|---|---|---|---|
| | | mean | median ($50^{th}$ %ile) | $95^{th}$ %ile |
| waste emplaced > 3 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 85.9 | 1.43e-5 | 209 |
| waste emplaced > 5 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 437 | 0.00264 | 1710 |
| waste emplaced > 10 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 14400 | 113 | 81400 |

* - Results based on 5,000 simulations of the Model

Groundwater concentrations for all other radionuclides are much less than their respective GWPLs, with the exception of $^{129}$I, which has never been detected in the DU waste proposed for disposal at Clive.

The dose and groundwater concentration results indicate that the downward pathway is dominated by groundwater concentrations of $^{99}$Tc, whereas, the upward pathway is dominated by dose from radon. A trade-off is indicated in terms of DU waste placement. The lower the DU waste is placed, particularly the $^{99}$Tc contaminated DU waste, the greater the groundwater concentrations of $^{99}$Tc, but the lower the doses. Conversely the higher the DU waste is placed in the embankment, the lower the $^{99}$Tc groundwater concentrations, and the greater the dose to ranch workers. However, there is a wide range of DU waste configurations in the CAS embankment that satisfy both dose and groundwater performance objectives.

In addition to the individual dose assessments, the structure of the model allows population dose to be tracked. In keeping doses as low as reasonably achievable (ALARA) estimated dose to the entire population of individuals over time is needed. One such calculation is the cumulative dose to all ranch workers, hunters, and OHV enthusiasts, summed across all individuals and all years of the 10,000-yr simulation. These cumulative population doses, as TEDE, are shown in Table ES-4, considering the various cases of waste placement and whether the gully screening calculation is included in the analysis.

The population doses presented in Table ES-4 are very small. This is because the populations of receptors are small, and the individual doses that they might receive are small. Both NRC and DOE have suggested ALARA-based costs of $1,000 (without discounting) and $2,000 (with discounting) per person rem. With costs like these, the total ALARA costs are negligible compared to the cost of waste operations and disposal.

**Table ES-4. Peak cumulative population TEDE: statistical summary**

| | Peak population TEDE (rem) within 10,000 yr* | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | 35.2 | 29.2 | 87.3 |
| no gullies; waste > 5 m below cover | 4.07 | 3.46 | 9.78 |
| no gullies; waste > 10 m below cover | 0.0434 | 0.0356 | 0.103 |
| with gullies; waste > 3 m below cover | 378 | 172 | 1430 |
| with gullies; waste > 5 m below cover | 4.46 | 3.7 | 10.7 |
| with gullies; waste > 10 m below cover | 0.0448 | 0.0364 | 0.108 |

* - Results based on 5,000 simulations of the Model

This simple ALARA analysis is consistent with the inhospitable environment and the remoteness of the Clive facility, and confirms the findings of the individual dose assessment. ALARA is intended to support evaluation of options to reduce doses in a cost-effective manner, however, given the results of this ALARA analysis, it is not clear that further reduction in risk (dose) is necessary. It is important to realize that the ALARA analysis depends on the Model structure, specification and assumptions, and that it focuses on a specific aspect of a more complete benefit-cost or decision analysis. However, the results are otherwise compelling.

The final set of analyses that are important are the deep-time analyses. As described above, the deep-time model is very conservative in many ways with respect to dispersal of the DU waste material. Large lakes that obliterate the CAS embankment are assumed to return periodically, but the models of dispersion of the waste are very constraining.

Given the model, peak mean concentrations of 238U in lake water and sediment for the next 100 ky are presented in Tables ES-5 and ES-6. These results simply show the concentrations that might occur in response to obliteration of the site, and subsequent dispersal of the waste in a relatively confined system. The concentrations presented would decrease with each lake and climate cycle as more sediment is deposited with each lake event, and each lake event allows the remnants of the DU waste to be dispersed ever further afield.

**Table ES-5. Statistical summary of peak mean uranium-238 concentrations in lake water within the first 100-ky climate cycle**

| | Peak mean lake water concentration of uranium-238 within 100 ky (pCi/L) | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | 0.18 | 0.0010 | 1.1 |
| no gullies; waste > 5 m below cover | 0.17 | 0.0009 | 1.0 |
| no gullies; waste > 10 m below cover | 0.18 | 0.0009 | 1.3 |

* - Results based on 5,000 simulations of the Model

**Table ES-6. Statistical summary of peak mean uranium-238 concentrations in sediment within the first 100-ky climate cycle**

| | Peak mean sediment concentration of uranium-238 within 100 ky (pCi/g)* | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median (50th %ile)** | **95th %ile** |
| no gullies; waste > 3 m below cover | 1,600 | 1,300 | 3,600 |
| no gullies; waste > 5 m below cover | 1,500 | 1,300 | 3,400 |
| no gullies; waste > 10 m below cover | 1,500 | 1,300 | 3,400 |

* - Results based on 5,000 simulations of the Model

The quantitative results are summarized in Table ES-7. Doses are always less than 500 mrem in a year, and doses to the offsite receptors are always much less than 25 mrem in a year. Groundwater concentrations of $^{99}$Tc are always less than its GWPL except when the $^{99}$Tc contaminated waste is disposed below grade. Even in this case, the median groundwater concentration is only 113 pCi/L.

**Table ES-7. Summary of the results of the Clive DU PA Model**

| | without gullies: top of waste at | | | with gullies: top of waste at | | |
|---|---|---|---|---|---|---|
| **performance objective** | **3 m** | **5 m** | **10 m** | **3 m** | **5 m** | **10 m** |
| Dose to MOP below regulatory threshold of 25 mrem/year | Yes | Yes | Yes | Maybe[1] | Yes | Yes |
| Dose to IHI below regulatory threshold of 500 mrem/year | Yes | Yes | Yes | Yes | Yes | Yes |
| Groundwater maximum concentration of $^{99}$Tc in 500 years < 3790 pCi/L[3] | Yes | Yes | No[2] | Yes | Yes | No[2] |
| ALARA average total population cost equivalent over 10,000 years | $35,000 | $4,000 | $43 | $378,000 | $4,500 | $45 |

The results overall suggest clearly that there are disposal configurations that can be used to dispose of the quantities of DU included in the Model that are adequately protective of human health and the environment.

# 1.0 Background

One of the responsibilities of the Nuclear Regulatory Commission (NRC) is to ensure the safe disposal of commercially generated low-level radioactive waste. Non-defense-related depleted uranium (DU) waste falls under the jurisdiction of NRC, and requires a disposal option that is protective of human health and the environment. NRC currently regulates the disposal of DU waste as a low-level radioactive waste, in cooperation with "Agreement States". The Energy*Solutions* low-level radioactive waste disposal facility at Clive, Utah is a candidate for disposal of DU waste, and Utah is an Agreement State that has regulatory authority to determine if such disposal can occur in compliance with Utah and NRC regulatory requirements.

Adequate protection of human health and the environment is evaluated by conducting a Performance Assessment (PA). A PA is used to model potential transport of radionuclides from the disposed inventory to the accessible environment, and to estimate radiation dose to potential human receptors. The estimated doses are compared to performance objectives, which are specified as dose limits. If the estimated doses are less than the performance objectives, then adequate protection of human health has been demonstrated.

The purpose of this report is to present the results of a the Clive DU PA Model v1.0 (the Model), a computer model developed to inform performance assessment (PA) for disposal of some specific DU waste materials at the Clive Facility. This report provides a summary of the approach taken and the results that can be obtained from the Model, and is accompanied by supporting documentation that includes details of the Model development and quality assurance program.

## 1.1 Depleted Uranium

In order to produce suitable fuel for nuclear reactors and/or weapons, uranium has to be enriched in the fissionable $^{235}U$ isotope. Uranium enrichment in the US began during the Manhattan Project in World War II. Enrichment for civilian and military uses continued after the war under the U.S. Atomic Energy Commission, and its successor agencies, including the DOE.

The uranium fuel cycle begins by extracting and milling natural uranium ore to produce "yellow cake," which is a varying mixture of uranium oxides. Low-grade natural ores contain about 0.05 to 0.3% by weight of uranium oxide while high-grade natural ores can contain up to 70% by weight of uranium oxide. Uranium found in natural ores contains two principal isotopes – uranium-238 (99.3% $^{238}U$) and uranium-235 (0.7% $^{235}U$). The uranium is enriched in $^{235}U$ before being made into nuclear fuel, which generates a product consisting of 3% to 5% $^{235}U$ for use as nuclear fuel and a by-product of DU (between 0.1% and 0.5 $^{235}U$). The DU has some commercial applications including counterweights and military applications as artillery. However, the commercial demand for depleted uranium is currently much less than the amounts generated for nuclear fuel. Use of $^{238}U$ as fuel for breeder reactors has not been seriously considered in this country. The U.S. Department of Energy (DOE) has about 700 Gg (700,000 Mg or metric tons) of DU in storage. Hence, the need to find disposal options for DU waste.

## 1.2 The Clive Waste Disposal Facility

Energy*Solutions* operates a low-level radioactive waste disposal facility west of the Cedar Mountains in Clive, Utah, as shown in Figure 1. Clive is located along Interstate-80, approximately 5 km (3 mi) south of the highway, in Tooele County. The facility is approximately 80 km (50 mi) east of Wendover, Utah and approximately 100 km (60 mi) west of Salt Lake City, Utah. The facility sits at an elevation of approximately 1302 m (4275 ft) above mean sea level (amsl) and is accessed by both road and rail transportation.

Currently, the Clive Facility receives low-level radioactive waste shipped via truck and rail. The Clive disposal facility is licensed to accept Class A low-level radioactive waste. Under current NRC regulations, DU waste is considered Class A waste, in which case the Clive site is an option for disposal. However, NRC and the State of Utah are currently considering options for updating their regulations and rules (10 CFR 61 for NRC, and UAC R313-25-8(2) for the State of Utah), which is likely to force the requirement of a PA for disposal of DU. Pending the findings of the Clive DU PA, DU waste will be disposed in an above-ground engineered disposal embankment that is clay-lined with a composite clay and rock cap. The disposal embankment is designed to perform for a minimum of 500 years based on requirements of 10 CFR 61.7, and hence provides a possible solution for the long-term disposal of DU.

Clive is a remote and environmentally inhospitable area. Human activity at Clive has, historically, been very limited. The regulations (10 CFR 61 and Utah regulations R313-25-8) indicate the need to evaluate performance with respect to members of the public and inadvertent human intruders. However, the difference between these two categories of human receptors is somewhat blurred because of the types of human activities that are reasonable to consider in the general area of the disposal facility. These two categories of receptors are described further below in the context of the regulatory context of the Clive DU PA.

## 1.3 Regulatory Context

Energy*Solutions* is permitted by the State of Utah to receive Class A Low Level under Utah Administrative Code (UAC) R313 25, *License Requirements for Land Disposal of Radioactive Waste.* The wastes that are received must be classified in accordance with the UAC R313 15 1008, *Classification and Characteristics of Low-Level Radioactive Waste.* The classification requirements in UAC R313-15-1008 reflect those outlined in NRC's 10 CFR 61 Section 55, but include additional references to radium 226 ($^{226}$Ra). Further, groundwater protection levels (GWPLs) must be adhered to, as outlined in the site's *Ground Water Quality Discharge Permit* (UWQB, 2010).

Title 10 CFR 61 (Code of Federal Regulations, 2007) is the Federal regulation for the disposal of certain radioactive wastes, including land disposal at privately-operated facilities such as that managed and operated by Energy*Solutions* at Clive, Utah. It contains procedural requirements, performance objectives, and technical requirements for near-surface disposal, including disposal in engineered facilities with protective earthen covers, which may be built fully or partially above-grade. Near-surface disposal is defined as disposal in or within the upper 30 m (100 ft) of the earth's surface (10 CFR 61.2).


**Figure 1. Location of the Clive site operated by Energy*Solutions* (base image from Google Earth).**

Performance objectives are evaluated by preparing a PA model. DU presents an interesting case because the uranium is nearly all $^{238}$U, meaning that secular equilibrium is not attained for more than 2 My, and during that time, activity associated with the DU continues to increase. At the time of the development of the regulation, DU waste as such did not, and was not expected to, exist in significant quantities. The nature of the radiological hazards associated with DU presents challenges to the estimation of long-term effects from its disposal. Recognition of this special behavior of DU has prompted the NRC to revisit the regulation. Until that process is complete, however, 10 CFR 61 stands as the controlling regulation.

The key endpoints of a PA are estimated future potential doses to members of the public (MOP). The performance objectives specified in Subpart C of 10 CFR 61 are in the following section:

> **§ 61.41 Protection of the general population from releases of radioactivity.** Concentrations of radioactive material which may be released to the general environment in ground water, surface water, air, soil, plants, or animals must not result in an annual dose exceeding an equivalent of 25 millirems [0.25 mSv] to the whole body, 75 millirems [0.75 mSv] to the thyroid, and 25 millirems

> [0.25 mSv] to any other organ of any member of the public. Reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable.

The location of a member of the public (MOP) is not defined clearly in the NRC statute. Under DOE Order 435.1 the MOP is defined as someone who does not access the disposal facility, but is located outside of the fence line or boundary of the facility. However, NRC does not similarly define an MOP, unless the disposal facility is not considered part of the natural environment. Otherwise, an MOP is not restricted other than through the activities in which the MOP might engage.

In addition to addressing MOP, 10 CFR 61 requires additional assurance of protecting individuals from the consequences of inadvertent intrusion. An inadvertent intruder is someone who is exposed to waste without intent, and without realizing that exposure might occur (after loss of institutional control). This is distinct from the intentional intruder, who might be interested in deliberately disturbing the site, or extracting materials from it, or who might be driven by curiosity or scientific interest. Intentional intruders are not evaluated in a PA.

> **§ 61.42 Protection of individuals from inadvertent intrusion.** Design, operation, and closure of the land disposal facility must ensure protection of any individual inadvertently intruding into the disposal site and occupying the site or contacting the waste at any time after active institutional controls over the disposal site are removed.

The distinction between MOP and an inadvertent intruder is clear in DOE Order 435.1, but is not as clear in NRC 10 CFR 61. Under DOE Orders, a MOP does not engage in activities within the boundaries of the disposal facility, and an inadvertent intruder inadvertently accesses the waste material directly. Consequently, the locations of MOP and intruder are different under DOE Orders. However, the NRC indicates that an inadvertent intruder is defined as follows:

> **§ 61.2 Definitions**. *Inadvertent intruder* means a person who might occupy the disposal site after closure and engage in normal activities, such as agriculture, dwelling construction, or other pursuits in which the person might be unknowingly exposed to radiation from the waste.

Because of the remoteness of the Clive Facility and, hence, the types of activities in which humans might engage, the distinction is made for this PA that ranchers, hunters and OHV enthusiasts are inadvertent intruders because they "engage in normal activities, such as agriculture, dwelling construction, or other pursuits in which the person might be unknowingly exposed to radiation from the waste". This facility is regulated under NRC, in which case the definitions in 10 CFR 61 are most relevant. However, it is noted that the ranchers, hunters and OHV enthusiasts do not intrude into the waste to create a direct exposure. Other receptors evaluated in the PA Model who are located offsite are regarded as MOPs. The results of this Model are calculated without regard for MOP and IHI categorization. The Model simply evaluates dose to each receptor, providing the information necessary for comparison with performance objectives.

No dose limit is specified in 10 CFR 61 for the inadvertent intruder. However, since Part 61 has been issued, the standard used by NRC and others for LLW disposal licensing has been an annual dose of 500 mrem. The 500 mrem-in-a-year standard is also used in the DOE waste determinations implementing the Part 61 performance objectives (NUREG-1854), and as part of the license termination rule dose standard for intruders (10 CFR 20.1403).

The scope of a PA may be limited to the evaluation of MOP and inadvertent intrusion, and also to the issue of site stability. The performance standard for stability requires the facility to be sited, designed, and closed to achieve long-term stability to eliminate to the extent practicable the need for ongoing active maintenance of the site following closure. The intent was to provide reasonable assurance that long-term stability of the disposed waste and the disposal site will be achieved. To help achieve stability, the NRC suggested to the extent practicable that disposed waste should maintain gross physical properties and identity over 300 years, under the conditions of disposal, with a further suggestion that the disposal facility should be evaluated for at least a 500-year time frame. About the same time as Part 61 was promulgated, the NRC also put in place requirements for design of uranium mill tailings piles such as the Vitro site which is collocated with the Clive Facility. The NRC specified that the design shall provide reasonable assurance of control of radiological hazards to be effective for 1,000 years to the extent reasonably achievable, and, in any case, for at least 200 years.

This raises the issue of appropriate compliance periods for a waste form that does not reach peak radioactivity for more than 2 My. Section 2(a) of R313-25-8 states:

> For purposes of this performance assessment, the compliance period shall be a minimum of 10,000 years. Additional simulations shall be performed for the period where peak dose occurs and the results shall be analyzed qualitatively.

The intent of this Model, therefore, is to evaluate impacts to receptors for a period of 10,000 years, and long-term performance of the disposal system beyond that time. The regulation does not address time frame for site stability. Given the long period of time before DU reaches secular equilibrium, it is difficult to determine when peak dose might occur. Consequently, the Clive DU PA Model has been implemented quantitatively for 10 ky, and has run additional simulations for 2.1 My, the time at which DU reaches peak activity. The results of the PA Model will be used to inform decisions about the suitability of the Clive facility for disposal of DU waste, the amount of DU waste that can be disposed safely, and different options for the engineered design and the placement of the waste within the disposal system. These decisions will be made in light of the doses to the receptors identified for the Model, groundwater concentrations of $^{99}$Tc and other radionuclides, and the long-term effects on site stability and dispersal of DU waste in returning lakes and lake sediment.

Site stability might also be considered to be a qualitative criterion for evaluating the concept of maintaining receptor impacts to be "as low as reasonably achievable" (ALARA). However, the CFR (Section 61.42) also defines ALARA in the context of dose to populations. The regulation states that "reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable". The ALARA process is described in more detail in the white paper *Decision Analysis Methodology for Assessing ALARA Collective*

*Radiation Doses and Risks* (Appendix 12). ALARA is evaluated in terms of population doses for the design options that are considered. This allows design options to be compared, and, ultimately, to be optimized. NRC also offers options for discounting costs of human exposures over time. NRC suggests a value of $2000 for the cost per person rem, with a possible range of $1000 to $6000. This range will be considered in the ALARA analysis. The ALARA analysis complements the compliance analysis for MOP and inadvertent intruders, since only those options that are in compliance are considered.

In addition to the radiological criteria, the State of Utah imposes limits on groundwater contamination, as stated in the Ground Water Quality Discharge Permit (UWQB, 2010). Part I.C.1 of the Permit specifies that GWPLs in Table 1A of the Permit shall be used for the Class A LLW Cell. Table 1A in the Permit specifies general mass and radioactivity concentrations for several constituents of interest to DU waste disposal. These GWPLs are derived from Ground Water Quality Standards listed in UAC R317-6-2 *Ground Water Quality Standards*. Exceptions to values in that table are provided for specific constituents in specific wells, tabulated in Table 1B of the Permit. This includes values for mass concentration of total uranium, radium, and gross alpha and beta radioactivity concentrations for specific wells where background values were found to be in exceedence of the Table 1A limits.

According to the Permit, groundwater at Clive is classified as Class IV, saline ground water, according to UAC R317-6-3 *Ground Water Classes,* and is highly unlikely to serve as a future water source. The underlying groundwater in the vicinity of the Clive site is of naturally poor quality because of its high salinity and, as a consequence, is not suitable for most human uses, and is not potable for humans. However, the Clive DU PA Model calculates estimates of groundwater concentrations at a virtual well near the Class A South Cell for comparison with these GWPLs. Part I.D.1 of the Permit specifies that the performance standard for radionuclides is 500 years.

## 1.4 Performance Assessment

Within the regulatory framework described above, a PA addresses doses to potential human receptors within a time frame of compliance. The Clive DU PA Model also addresses performance of the system for approximately 2.1 My—until secular equilibrium of $^{238}$U and its decay products is reached. The PA process starts with the regulatory context, but is itself a decision support process. Decisions may be made based on the results of the PA modeling that is performed. In the context of decision analysis, this requires steps that include:

1. State a problem,
2. Identify objectives (and measures of those objectives – i.e., attributes or criteria),
3. Identify decision alternatives or options,
4. Gather relevant information, decompose and model the problem (structure, uncertainty, preferences),
5. Choose the "best" alternative (the option that maximizes the overall benefit),

6. Conduct uncertainty analysis, sensitivity analysis and value of information analysis to determine if the decision should be made, or if more data/information should be collected to reduce uncertainty and, hence, increase confidence in the decision, and
7. Go back (iterate) if more data/information are collected.

The problem addressed here is one of potential disposal of DU waste at the Clive Facility. The objectives are to minimize risk to human health and the environment. Risk is measured in terms of dose and uranium toxicity hazard to the human receptors that are identified for analysis. The decision options that are evaluated relate to different waste configuration options for DU waste disposal. Given that context, the next step of the PA process is to gather information, and build a PA model. There are several steps involved, each one building on the previous step. The modeling process starts with evaluating features, events and processes (FEPs) that might be important for evaluating performance, and using the FEPs analysis to build a conceptual site model (CSM). These steps are described in full in the *FEP Analysis for Disposal of Depleted Uranium at the Clive Facility* (Appendix 1), and the *Conceptual Site Model for Disposal of Depleted Uranium at the Clive Facility* (Appendix 2).

Development of the CSM sets the stage for subsequent model structuring, which is the first step needed to build the numerical model of the system. All relevant FEPs are captured in the model structure, from waste inventory, mechanisms for transport through the engineered system, migration through the natural environment to the accessible environment, to identification of human receptors, exposure pathways and dose assessment. The model structure leads to specification of the model. Probability distributions are specified for each input parameter. The type of information available for each input parameter is highly variable, hence requiring varied approaches for specification. Different methods that are used are described in the white paper *Development of Probability Distributions* (Appendix 14).

Model structuring and specification completes the numerical model. The model is computed using the GoldSim systems analysis software (GTG, 2010). GoldSim is probabilistic simulation software that includes a graphical user-interface that is convenient for developing PA models. GoldSim is inherently a systems-level software framework. The focus of a GoldSim model is on the decision making process, which includes managing uncertainty and coupling all processes. This 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 development of the model is iterative, where the iterations depend on model evaluation, which is performed at various levels. During model construction the model is evaluated iteratively as new components are added. Once a complete model is assembled then the model is subjected to uncertainty and sensitivity analysis. The goals of the uncertainty analysis are to evaluate results against the performance objectives and to understand the values of the results with respect to the model formation. The sensitivity analysis is used to identify components of the model that are most influential on the output. This leads to model iteration as suggested in Step 7 above.

Building a model to inform PA is a large undertaking. There are many intricacies that must be accommodated starting with development of FEPs, moving through the CSM, mathematical abstraction of environmental processes, numerical model structuring, development of probability distributions for the input parameters, and model evaluation. This complex process is described briefly in this document, and is described in more detail in the supporting documents (see Appendices). In addition to complete documentation, the GoldSim model itself is fully contained, with internal documentation of every aspect of the model structure. The extensive documentation is provided for two reasons: The first is simply that it provides access to all information used in the Model. This is done in the spirit of openness, transparency and, hence, defensibility. The second is in the context of the quality assurance program that requires tracking of all information from its source through to the final model. The QA program implemented for this Model is described in full in the *Quality Assurance Project Plan* (Appendix 17).

## 1.5 Technical Evolution of PA and PA Modeling

Since PA modeling began in the late 1970s through early 1990s at many of the radioactive waste disposal facilities around the U.S., many different approaches to modeling have been used. These approaches span the range from deterministic process-level modeling to probabilistic systems-level modeling. Early PA models tended towards deterministic modeling for several reasons: 1) PA modeling was initially performed with a focus on groundwater modeling, which was, and still is, often performed using deterministic process-level models, 2) there were computational or technological difficulties with taking a probabilistic approach, and 3) PA regulations and guidance were established mostly with deterministic performance objectives, which was interpreted as a reason for performing deterministic modeling. In particular, PA for low-level radioactive waste (LLW) disposal facilities followed deterministic performance objectives. However, the regulations for the Waste Isolation Pilot Plant and the Yucca Mountain Project (YMP) (Title 40, *Code of Federal Regulations* (CFR), Part 191, "Environmental Radiation Protection Standards for Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Wastes," and Title 40, CFR Part 197, "Public Health and Environmental Radiation Protection Standards for Yucca Mountain, Nevada") provide an exception to the deterministic objectives, and consequently, PA models for these radioactive waste disposal facilities have been developed probabilistically.

Technological advances in the last decade have also allowed more PA modeling to move towards a probabilistic approach. Finally, PA modeling is multi-disciplinary, and as more technical disciplines have been brought into PA modeling, there has been increased recognition of the potential benefits of probabilistic systems-level modeling.

Systems-level models are usually computationally simpler than process-level models. However, the systems-level PA model might still have large numbers of parameters, which reveals the complexity of dealing with PA modeling even at a systems-level scale. The large number of parameters is a consequence of the many constituents of concern that are usually included in PA models, and the need to characterize transport properties for each of these constituents (e.g., partitioning coefficients, solubility, plant uptake factors). However, it is unlikely that more than a few of these parameters are important predictors for a given PA endpoint (e.g., dose to a member of the public, groundwater protection levels). Along these lines, another advantage of systems-level modeling performed in a probabilistic environment is the ability to identify parameters that are most important or sensitive for a given endpoint. Because system-level models may be probabilistic, global sensitivity analysis methods can be used to identify the most sensitive parameters (see the white paper entitled *Sensitivity Analysis* in Appendix 15).

The advantages of system-level models are that they are capable of 1) coupling of different processes without the need for the application of ad hoc boundary conditions, 2) using an appropriate spatial and temporal scaling relative to the decisions that need to be made, 3) having the ability to characterize and manage uncertainty through probabilistic modeling, and 4) being used to perform global sensitivity analysis. Use of the global sensitivity analysis can potentially lead to refinement and enhancements of the underlying models or the identification and collection of new data (e.g., research studies or monitoring) as necessary to reduce uncertainty of certain parameters or variables. Use of a system-level model can also provide the ability to rapidly and efficiently explore alternative conceptualizations of the system, which allows a greater ability to address scenario and conceptual model uncertainties.

System-level models are often supported by process-level models. Each component of a system-level model requires model building, which can include abstraction from a process-level model. The purpose of the abstraction is to be able to capture the essence of the process-level model in the probabilistic system-level model, so that its relative importance or sensitivity can be evaluated. As a consequence of the development of system-level modeling frameworks such as GoldSim, PA models are often developed following this approach, with global sensitivity analysis driving iteration until the model results indicate a clear response and decision path.

## 1.6 Report Structure

The remainder of this report provides a more complete introduction to the PA modeling process applied to the Clive DU waste disposal option, briefly describes the FEPs process, and follows with a brief description of the CSM. The CSM description is aimed more at identifying components of the model that might be significant in the model results. Model building always leads to insights into the important components of a model, and that is conveyed in terms of important aspects of the CSM.

The model structure is described prior to presentation of results, which are the main focus of this report. Results are presented for the 10-ky quantitative model and for the deep-time model. For the 10-ky model, the important results from a regulatory perspective include doses to the receptors that have been identified as critical. Groundwater concentrations are evaluated for the next 500 yrs. For the deep-time model, which models the performance of disposal of DU at Clive for the next 2.1 My, results are presented in terms of lake water concentrations assuming the return of a large pluvial lake in the Bonneville Basin, and sediment concentrations that remain after the pluvial lake recedes.

A summary is provided that includes further interpretation of results and comparison with performance objectives. More complete documentation of the details of the model development is contained in the Appendices, and also in the GoldSim model itself. This compendium of documents provides a thorough treatise of the Clive DU PA Model v1.0.

# 2.0 Introduction

The safe storage and disposal of DU waste is essential for mitigating releases of radioactive materials and reducing exposures to humans and the environment. Currently, a radioactive waste facility located in Clive, Utah and operated by Energy*Solutions* is proposed to receive and store DU waste that has been declared surplus from radiological facilities across the nation. The Clive Facility has been tasked with evaluating disposal of the DU waste in an economically feasible manner that protects humans from future radiological releases.

To assess whether the 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 Title 10 Code of Federal Regulations Part 61 (10 CFR 61) Subpart C, and promulgated by the Nuclear Regulatory Commission (NRC), must be met. In order to support the required radiological PA, a model is needed to evaluate doses to human receptors that would result from the disposal of DU and its associated radioactive contaminants.

This section provides an introduction to the general approach taken to developing version 1.0 of the Clive DU PA Model. The focus is on methods that have been undertaken at each step along the path, from description of the problem and the disposal facility under consideration, FEPs identification, CSM development, approaches to numerical modeling and evaluation of results.

## 2.1 General Approach

Performance Assessment models are complex probabilistic systems-level models that evaluate the long-term effects to human health and the environment of disposal of radioactive waste. The approach includes the following steps:

1. Identification of disposal options – in this case use of the Class A South embankment at the Clive Facility in Utah for disposal of DU waste, and specifics of the disposal configuration. This includes consideration of the regulatory environment in which the PA model is to be evaluated.

2. Identification of important FEPs that should be considered in the evaluation of the Clive disposal facility. This includes identification of human receptors who might be engaged in activities near or on the disposal facility.

3. Development of a CSM that captures the relevant FEPs. This includes cursory evaluation of the FEPs for the likelihood of occurrence and their consequence. If, for a given FEP the likelihood of occurrence or consequence is considered too small, then the FEP is not included in the CSM.

4. Development of a numerical or computational model for the PA. This translates the CSM into numerical code for processing. This includes model structure and model specification. The Clive DU PA Model is developed fully probabilistically, with coupling of all processes included in the model.

5. Model evaluation, including:
   a. uncertainty analysis, which compares the probabilistic output to the performance objectives,
   b. sensitivity analysis, which is used to identify the important parameters or components of the model in terms of prediction of the model output. This leads to model refinement or data collection if the uncertainties in the decisions that need to be made are considered to be too large.
6. Reporting of the PA model and its results, including:
   a. Doses to potential human receptors
   b. Population doses evaluated in the context of ALARA
   c. Groundwater concentrations
   d. Deep time concentrations in lake water and lake sediment
7. Quality Assurance.

A PA is a type of systematic (risk) analysis that addresses (a) what can happen, (b) how likely it is to happen, (c) what the resulting impacts are, and (d) how these impacts compare to regulatory standards. The essential elements of a performance assessment are (a) a description of the site and engineered system, (b) an understanding of events and processes likely to affect long-term facility performance, (c) a description of processes controlling the movement of contaminants from waste sources to the general environment, (d) a computation of metrics reflecting system performance including concentrations, doses, and other human health risk metrics to members of the general population, and (e) an evaluation of uncertainties in the modeling results that support the assessment.

Because of the long-term nature of the analysis, the intent of a PA is not necessarily to estimate actual long-term human health impacts or risks from a closed facility. Rather, the purpose of the Model is to provide a robust analysis that can examine and identify the key elements and components of the site, the engineered system, and the environmental setting that could contribute to potential long-term impacts. Because of the time-scales of the analysis and the associated uncertainty in knowledge of characteristics of the site, the waste inventory, the engineered system and its potential to degrade over time, and changing environmental conditions, a critical part of the PA process is also the consideration of uncertainty and evaluation of model and parameter sensitivity in interpretation of PA modeling results.

A probabilistic model includes a mathematical analysis of stochastic events or processes and their consequences. Probabilistic analysis acknowledges that events and processes are inherently uncertain, and hence involves characterization of uncertainty around expectation. Model output hence is expressed with the same characteristics of expectation and uncertainty, which lends itself to a global or probabilistic sensitivity analysis. Sensitivity analysis for probabilistic models is used to identify the parameters (variables) that are the most important predictors of the output for a given endpoint (e.g., dose to a resident, concentrations in groundwater). The important predictors are those that explain most of the variability in the output variable of interest. Usually, for a given endpoint of interest, this is no more than a handful of input or explanatory variables. Because PA models are usually complex, dynamic, non-linear systems, these global sensitivity analysis methods involve complex non-linear regression models that capture the input of each input variable across its specified range (range of its probability distribution).

PA concerns modeling radioactive waste disposal facilities into the long-term future. As such, PA models must address both the spatial and temporal magnitude of PA. It is critical in a PA model to addresses the scale of the decisions that need to be made. Modeling is performed at the spatial and temporal scale that is needed to support PA decisions related to closure. In effect, system-level models might be fairly coarse, but this has advantages for evaluating how the system evolves over time. For example, all processes involved are fully coupled in the same model, probabilistic modeling can be performed to both characterize and manage uncertainty, and statistics and decision analysis can be incorporated into the modeling framework.

Results from a systems-level model are aimed at the decision objectives at the spatial and temporal scales of interest. These results are presented as probability distributions for the endpoints of interest (doses, concentrations, etc.), and comparisons are made with performance objectives where appropriate (dose, groundwater concentrations).

Given the PA model construction with respect to the spatio-temporal scales of the model, there are two levels of response. The first is for each hypothetical individual included in the model. Dose results are available for each receptor in every year of the model, up to 10 ky. Each dose result at this level represents individual dose to the concentrations in various media predicted by the model at that time. The dose parameters, however, are specific to the individual. This approach to modeling dose was taken for a few reasons: 1) There are not many receptors at Clive, in which case, from a computational perspective it was feasible to consider each individual receptor, and 2) This approach allows population dose to be estimated directly from the individual doses.

Although individual doses are available in the model, the output of interest is the mean dose. Traditionally this has been estimated as the mean dose to a hypothetical average individual. With this model, the mean dose is estimated directly from the individual doses. Mean doses are evaluated in each year of the model, however, traditionally for PA, interest lies primarily in the worst case year, in which case the peak mean dose across time is the metric of interest.

The effect is that average (mean) doses are available at multiple scales. Traditional comparison with performance objectives is performed with the peak mean dose, meaning the highest mean dose in a year across the 10-ky performance period. This simplification might have been taken previously because of technical practicability. However, with modern computer technology, such short-cuts are not necessary, and the mean dose within each model year can be evaluated directly. However, in the interest of precedent, the "peak of the means" is used in this document for comparison purposes. The problem with the peak of the means is that the peak might vary in time form simulation to simulation. Considering the peak of the means in this way overestimates dose, and, consequently, underestimates disposal system performance. In this model, for which radioactivity is increasing with time for the DU waste, the peak almost always occurs close to 10 ky, in which case this is not a major issue. The distribution of the peak of the means is presented in this report. Note that there are 5,000 estimates of the peak of the mean for each receptor from the 5,000 simulations that are run. This is usually enough simulations to stabilize an estimate of the mean. The dose assessment model is described in detail in the white paper entitled *Dose Assessment* (Appendix 11).

If the distribution of the peak of the means is treated as if each simulation result is independent, then, because the model is constructed at the spatial and temporal scales as described above, the 95$^{th}$ percentile of the distribution is somewhat analogous to the notion of a 95% upper confidence limit that is commonly used under CERCLA. Comparisons may be made with the PA performance objectives using the median, mean and 95$^{th}$ percentile of the output distribution for each endpoint of interest.

For the ALARA analysis, the model is set up so that the population dose can be estimated for each receptor class in each year of the model. The 5,000 realizations provide 5,000 estimates of population dose in each year of the model. The population dose distribution can also be processed to include the cost to human health and society by assigning a dollar value to person rem. This process is described in detail in the *Decision Analysis* white paper (Appendix 12).

Once the results are obtained and compared to the performance objectives, a global sensitivity analysis is performed to identify the parameters that are the most influential in predicting each endpoint of interest. Often this is only a handful of parameters for each endpoint. The results of the sensitivity analysis can be used to determine if it might be useful to collect more data or otherwise refine the model before making final decisions. This is ostensibly a decision analysis task, which can be performed using the sensitivity analysis results as a basis for determining the benefit of collecting new data. The potential benefits would be seen in reduction in uncertainty in the model results. The sensitivity analysis methods used for this model are described in the white paper entitled *Sensitivity Analysis Methods* (Appendix 15).

This holistic approach to PA modeling is aimed at providing insights into disposal system performance. Although the model predicts or estimates doses to human receptors, among other endpoints, the more important aspect of this type of modeling is to gain an understanding of how the system might evolve over the time frames of interest, and to use this understanding to support decision making including ability to safely dispose of waste and optimization of waste placement within the disposal system.. No matter what doses are predicted, it is important to understand why those modeled doses are observed, and hence, what are the important features of the disposal system with regards to protection of human health and the environment.

## 2.2 General Facility Description

The Energy*Solutions* low-level radioactive waste disposal facility is west of the Cedar Mountains in Clive, Utah, as shown in Figure 2. Clive is located along Interstate-80, approximately 5 km (3 mi) south of the highway, in Tooele County. The facility is approximately 80 km (50 mi) east of Wendover, Utah and approximately 100 km (60 mi) west of Salt Lake City, Utah. The facility sits at an elevation of approximately 1302 m (4275 ft) above mean sea level (amsl). The Clive Facility is adjacent to the above-ground disposal cell used for uranium mill tailings that were removed from the former Vitro Chemical company site in South Salt Lake City between 1984 and 1988 (Baird et al., 1990).

Currently, the Clive Facility receives waste shipped via truck and rail. Pending the findings of the PA, DU waste will be stored in a permanent above-ground engineered disposal embankment that is clay-lined with a composite clay and rock cap. The disposal embankment is designed to perform for a minimum of 500 years based on requirements of 10 CFR 61.7. The Energy*Solutions* Clive Facility is divided into three main areas (Figure 2):

- the Bulk Waste Facility, including the Mixed Waste, Low Activity Radioactive Waste (LARW), 11e.(2), and Class A LLW areas,
- the Containerized Waste Facility (CWF), located within the Class A LLW area, and
- the Treatment Facility (TF), located in the southeast corner of the Mixed Waste area.


**Figure 2. Disposal and Treatment Facilities operated by Energy*Solutions*.**

The DU waste under consideration is proposed for disposal in the Class A South (CAS) cell. The terms "cell" and "embankment" are here used interchangeably. That is, this Clive DU PA Model considers only to the long-term performance of DU disposed in this waste cell. The CAS embankment, or cell, is the western fraction of the Federal Cell (Figure 2). The eastern section is occupied by the 11e.(2) cell, which is dedicated to the disposal of uranium processing by-product waste, but not considered in this analysis.

The general aspect of the CAS embankment is that of a hipped cap, with relatively steeper sloping sides nearer the edges. The upper part of the embankment, known as the top slope, has a moderate slope, while the side slope is markedly steeper (20% as opposed to 2.4%). For this PA Model, no waste is placed under the side slopes, in which case modeling focuses on waste placed under the top slope. The embankment is also constructed such that a portion of it lies below-grade. Details of the design of the embankment are contained in the white paper entitled *Embankment Modeling* (Appendix 3).

DU waste from the Savannah River Site (SRS) and the gaseous diffusion plants (GDP) at Portsmouth, Ohio and Paducah, Kentucky has been proposed for disposal at the Clive facility. There are three categories of DU waste that are considered:

1. Depleted uranium oxide ($UO_3$) waste from the Savannah River Site (SRS) proposed for disposal at the Clive facility,
2. DU from the GDPs, which exists in two principal populations:
   a) DU contaminated with fission and activation products from reactor returns introduced to the diffusion cascades, and
   b) DU consisting of only "clean" uranium, with no such contamination.

The DU oxides that are to be produced at these sites "deconversion" plants will be primarily $U_3O_8$. The contamination problem arises from the past practice of introducing irradiated nuclear materials (reactor returns) into the isotopic separations process. Irradiated nuclear fuel underwent a chemical separation process to remove the plutonium for use in nuclear weapons. Uranium, then thought to be a rare substance, was also separated out, but contained some residual contamination from activation and fission products. This uranium was again converted to $UF_6$ for re enrichment, and was introduced to the gaseous diffusion cascades, contaminating them and the storage cylinders as well. Decay products ($^{226}$Ra), activation products ($^{241}$Am, $^{237}$Np, $^{238}$Pu, $^{239}$Pu, $^{240}$Pu, $^{241}$Pu, $^{242}$Pu), and fission products ($^{90}$Sr, $^{99}$Tc, $^{129}$I, $^{137}$Cs) potentially contaminate the DU waste. The proposed inventory that is evaluated in the Model is described fully in the white paper entitled *Waste Inventory* (Appendix 4).

# 3.0 Features, Events and Processes

The conceptual site model (CSM) describes the physical, chemical, and biological characteristics of the Clive facility. The CSM, therefore, encompasses everything from the inventory of disposed wastes, the migration of radionuclides contained in the waste through the engineered and natural systems, and the exposure and radiation doses to hypothetical future humans. These site characteristics are used to define variables for the quantitative PA model that is used to provide insights and understanding of the future potential human radiation doses from the disposal of DU waste.

The content of the CSM informs the Model with respect to regional and site-specific features, events and processes, such as climate, groundwater, and human receptor scenarios. The CSM accounts for and defines relevant features, events, and processes (FEPs) at the site, materials and their properties, interrelationships, and boundaries. These constitute the basis of the Model, on which, or through which, radionuclides are transported to locations where receptors might be exposed.

A key activity in developing a PA for a radiological waste repository is the comprehensive identification of relevant external factors that should be included in quantitative analyses. These factors, termed "features, events, and processes" (FEPs), form the basis for scenarios that are evaluated to assess site performance.

The universe of FEPs that were screened and identified as relevant for the Clive Facility PA are documented in the white paper entitled *FEP Analysis for Disposal of Depleted Uranium at the Clive Facility* (Appendix 1) and further elaborated in the CSM document (*Conceptual Site Model for Disposal of Depleted Uranium at the Clive Facility* – Appendix 2).

# 4.0 Conceptual Site Model

The important components of the conceptual site model are described in the following sections. Details are contained in the white paper entitled *Conceptual Site Model for Disposal of Depleted Uranium at the Clive Facility* (Appendix 2).

### 4.1.1 Disposal Site Location

Energy*Solutions* operates a low-level radioactive waste disposal facility west of the Cedar Mountains in Clive, Utah, as shown in Figure 1. Clive is located along Interstate-80, approximately 5 km (3 mi) south of the highway, in Tooele County. The facility is approximately 80 km (50 mi) east of Wendover, Utah and approximately 100 km (60 mi) west of Salt Lake City, Utah. The facility sits at an elevation of approximately 1,302 m (4,275 ft) above mean sea level (amsl) and is accessed by both highway and rail transportation. The Clive Facility is adjacent to the above-ground disposal cell used for uranium mill tailings that were removed from the former Vitro Chemical company site in South Salt Lake City between 1984 and 1988 (Baird et al., 1990).

### 4.1.2 Disposal Site Description

Currently, the Clive Facility receives waste shipped via truck and rail. DU waste is proposed for disposal in a permanent above-ground engineered disposal embankment that is clay-lined with a composite clay and rock cap. The disposal embankment is designed to perform for a minimum of 500 years based on requirements of 10 CFR 61.7, which provides a long-term disposal solution with minimal need for active maintenance after site closure. More detail relating to the properties of the disposal embankment is provided in Section 4.1.2.1.

The Energy*Solutions* Clive Facility is divided into three main areas (Figure 2): the Bulk Waste Facility, including the Mixed Waste, Low Activity Radioactive Waste (LARW), 11e.(2), and Class A LLW areas, the Containerized Waste Facility (CWF), located within the Class A LLW area, and the Treatment Facility (TF), located in the southeast corner of the Mixed Waste area. This analysis considers only the Class A South (CAS) embankment.

#### 4.1.2.1 Embankment

Depleted uranium waste is proposed for disposal in the Class A South disposal cell. The Class A South (CAS) Cell, which is part of the Federal Cell, is about 541 × 436 m (1,775 × 1,430 ft), with an area of approximately 24 ha (58 acres), and an estimated total waste volume of about 2.7 million $m^3$ (96 million $ft^3$). A drainage ditch surrounds the disposal cell on three sides, with 11e.(2) waste on the fourth side. The cell is constructed on top of a compacted clay liner covered by a protective cover. Waste will be placed above the liner and will be covered with a layered engineered cover constructed of natural materials. The top slopes will be finished at a 4% grade while the side slopes will be no steeper than 5:1 (20% grade).

The design of the Class A South Cell cover has been engineered to discourage erosion, reduce the effects of infiltration, and to protect workers and the public from radionuclide exposure. The cell cover is a layered composite of a clay radon barrier, filter material, sacrificial soil, and rip rap. The clay radon barrier is designed to minimize infiltration of precipitation and runoff and reduce the migration of radon from the waste cell. The filter material is intended to confine dew and condensates in order to reduce the likelihood of the radon barrier clay from drying out. The purpose of the rip rap cover is to ensure the integrity of the underlying layers and overall waste cell by providing protection from physical weathering sources such as erosion by water and wind. The detailed properties of each cell layer may be found in the white paper on *Embankment Modeling* (Appendix 3).

#### 4.1.2.2 Waste Inventory

The waste inventory is limited to the disposal of DU wastes of two general waste types: 1) depleted uranium trioxide ($DUO_3$) waste from the Savannah River Site (SRS) and 2) anticipated DU waste as $U_3O_8$ from gaseous diffusion plants (GDPs) at Portsmouth, Ohio and Paducah, Kentucky. The quantity and characteristics of DU waste from other sources that has that already been disposed of at the Clive Facility was not included. A full list of radionuclides has been established for the PA modeling effort. The radionuclide species list was based upon process knowledge, radionuclides analyzed for (though not necessarily detected) in the DU waste material, and decay products with half-lives over five years. The species list consists of the following radionuclides:

*fission products*:
Sr-90, Tc-99, I-129, Cs-137

*progeny of uranium isotopes*:
Pb-210, Rn-222, Ra-226, -228, Ac-227, Th-228, -229, -230, -232, Pa-231

*uranium isotopes*:
U-232, -233, -234, -235, -236, -238

*transuranic radionuclides*:
Np-237, Pu-239, -239, -240, -241, -242, Am-241

The waste inventory is discussed in more detail in the *Waste Inventory* white paper (Appendix 4) and in the *Conceptual Site Model* white paper (Appendix 1).

#### 4.1.2.3 Climate

The following sections briefly describe the aspects of the regional climate that influence the performance of the site and engineered features. Further details are provided in the *Conceptual Site Model* white paper (Appendix 1), and in the *Unsaturated Zone Modeling* white paper (Appendix 5). In general the climate is dry, with evapotranspiration potential that exceeds precipitation on an annual basis. This leads to low infiltration rates, and subsequent relatively slow movement of radionuclides to groundwater. Also, the embankment is largely above grade, and the dry, sometimes windy, environment could lead to drying out of the embankment beyond what is considered in typical unsaturated zone models.

##### 4.1.2.3.1 Temperature

Regional climate is regulated by the surrounding mountain ranges, which restrict movement of weather systems in the vicinity of the Clive facility. The most influential feature affecting regional climate is the presence of the Great Salt Lake, which can moderate downwind temperatures since it never freezes (NRC, 1993). The climatic conditions at the Clive Facility are characterized by hot and dry summers, cool springs and falls, and moderately cold winters (NRC, 1993). Frequent invasions of cold air are restricted by the mountain ranges in the area. Data from the Clive Facility from 1992 through 2009 indicate that monthly temperatures range from about -2°C (29°F) in December to 26°C (78°F) in July (Whetstone, 2006).

##### 4.1.2.3.2 Precipitation

The Clive Facility is characterized as being an arid to semi-arid environment where evaporation greatly exceeds annual precipitation (Adrian Brown, 1997). Data collected at the Clive Facility from 1992 through 2004 indicate that average annual rainfall is on the order of 22 cm (8.6 in) per year (Whetstone, 2006). Precipitation generally reaches a maximum in the spring (1992-2004 monthly average of 3.2 cm [1.25 in] in April), when storms from the Pacific Ocean are strong enough to move over the mountains (NRC, 1993; Whetstone, 2006). Precipitation is generally lighter during the summer and fall months (1992-2004 monthly average of 0.8 cm [0.32 in] in August) with snowfall occurring during the winter months (Whetstone, 2006; NRC, 1993; Baird et al., 1990).

#### 4.1.2.3.3 Evaporation

Because of warm temperatures and low relative humidity, the Clive Facility is located in an area of high evaporation rates. NRC (1993) indicates that average annual pond evaporation rate at the Clive Facility is 150 cm/yr (59 in/yr), with the highest evaporation rates between the months of May and October. Previous modeling studies indicate that the Dugway climatological station nearby is comparable to the Clive site with respect to evaporation and have reported pan-evaporation estimates of 183 cm/yr (72 in/yr), which is considerably greater than average annual rainfall (Adrian Brown, 1997). Because of the high evaporation rate, the amount of groundwater recharge due to precipitation is likely very small except during high intensity precipitation events (Adrian Brown, 1997).

### 4.1.2.4 Unsaturated Zone

The engineered features of the landfill, including cap, waste, and liner, are all in the unsaturated zone (UZ), at least within the 10,000-yr duration of the quantitative model. The part of the UZ that extends from the bottom of the cell liner to the water table consists of naturally-occurring lake sediments from the ancestral Lake Bonneville. Since the cap is intentionally designed to restrict permeability, interstitial water in the UZ below the facility is not expected to migrate upwards through the cap to surface soils, as it might otherwise do naturally given the strong evaporation potential at the surface. Rather, it is expected to migrate slowly down to the water table, at a rate equal to the rate at which the engineered liner leaks.

Diffusion in the water phase may also play a role in the transport of waterborne contaminants in the UZ, since the advective flux is expected to be small. The concentration gradients in the UZ are also expected to be predominantly vertical, so diffusion will also occur in the vertical direction, oriented with the column of cells.

Diffusion in the air phase within the UZ below the facility will not be modeled, since the only diffusive species would be radon, which is of greater concern at the ground surface. Upward radon diffusion to the ground surface will be dominated by radon parents in the waste zone, and is modeled within the engineered cap. Unsaturated zone processes, material properties, and parameters represented in the PA model are described in detail in the *Unsaturated Zone Modeling* White Paper. The primary concerns for the PA are movement through the unsaturated zone of mobile radionuclides, such as $^{90}$Sr, $^{99}$Tc, and $^{129}$I to groundwater and the upward diffusive movement of radon.

#### 4.1.2.4.1 Infiltration

The infiltration model for the cap and cell uses calculations from the HELP program to develop vertical and lateral flow rates in the individual layers of the cap. The results of the HELP modeling determine the vertical flow of water through the engineered cell layers, the waste, and the unsaturated zone. A numerical solution of Darcy's equation is used to determine the moisture contents in the radon barriers, waste layer, clay liner, and unsaturated zone from the vertical flow rates.

Comparisons of HELP modeling results with results from mechanistic unsaturated zone modeling programs such as UNSAT-H and HYDRUS at arid and semi-arid sites suggest that the HELP model will generally overestimate the vertical flow rates through waste cell covers (Meyer et al. 1996, Khire et al. 1997, Albright et al. 2002). These model comparisons indicate that the vertical flow rates through the CAS cell calculated using the HELP model are likely to be overestimated in the PA Model.

The cell and unsaturated zone infiltration modeling approaches and results are described in more detail in the *Unsaturated Zone Modeling* white paper (Appendix 5).

#### 4.1.2.5 Geochemical

The conceptual model for the transport of radionuclides at the Clive Facility allows sufficient meteoric water infiltration into the waste zone to allow dissolution of uranium and daughters, fission products and potential transuranic contaminants (along with native soluble minerals). At first, leaching is likely to be solubility-limited with respect to uranium, and the leachate will migrate away from the source with the uranium concentration at the solubility limit. The other radionuclides are unlikely to be at a solubility limit. Depending upon the amount of water available, these radionuclides will either re-precipitate, once the thermodynamic conditions for saturation are reached, or remain in solution and be transported to the saturated zone. This water is expected to be oxidizing, with circum-neutral to slightly alkaline pH (similar to the upper unconfined aquifer), and an atmospheric partial pressure of carbon dioxide. However, the amount of total dissolved solids (TDS) is expected to be initially lower than the upper aquifer.

The composition of this aqueous phase will change as it reaches the saturated zone, with some increase in dissolved solids and potentially lower dissolved oxygen and carbon dioxide. The saturated zone for this PA model includes only the shallow, unconfined aquifer. Transport of radionuclides is expected to be restricted to this aquifer and not migrate to the lower aquifer due to a natural upward gradient at the facility. The chemical composition of the saturated zone is characterized as somewhat alkaline pH likely due to the presence of carbonates, mainly oxidizing though transient reduced conditions may exist, with high levels of dissolved ions of mainly sodium and chlorine.

The transport of dissolved radionuclides can also be limited by sorption onto the solid phase of associated minerals and soils within each of the zones considered in this PA model. The transport of uranium is limited by both solubility and the sorption of radionuclides in groundwater. Sorption consists of several physicochemical processes including ion exchange, adsorption, and chemisorption. Sorption is represented in the PA model as a partitioning coefficient ($K_d$) value.

Distributions of radionuclide-specific partitioning coefficients and solubilities were developed for the PA model considering the geochemical conditions in the cell, the unsaturated zone, and the shallow aquifer at the Clive facility. The development of these distributions is described in detail in the *Geochemical Modeling* white paper (Appendix 6). The primary concerns for the model include the geochemical properties of $^{99}$Tc as they affect movement to groundwater, and of uranium in its different chemical forms for the 10-ky and deep-time models.

#### 4.1.2.6 Saturated Zone

Contaminants moving vertically in the UZ below the cell enter the saturated zone (SZ) beneath the disposal facility. The rate of recharge is the same as the Darcy flux (the rate of volume flow of water per unit area) through the overlying UZ, and is expected to be small enough that vertical transport within the SZ would be small. Most SZ waterborne contaminant transport will be in the horizontal direction, following the local pressure gradients, which are reflected in water table elevations in the shallow aquifer. A point of compliance in the groundwater has been established at 27 m (90 ft) from the edge of the embankment interior, so saturated transport is modeled to that point. Note that in the case of the proposed DU waste disposal, only the top slope section of the embankment would contain DU waste, so the effective distance from the DU waste to the well is lengthened by the width of the side slope section, to about 73 m (240 ft).

Saturated zone groundwater transport generally involves the processes of advection-dispersion and diffusion. Mean pore water velocity in the saturated zone is assumed to be determined by the Darcy flux and the porosity of the sediment. A range of values will allow the sensitivity analysis (SA) to determine if this is a sensitive parameter in the determination of concentrations at the compliance well and resultant potential doses. Modeling of fate and transport for the saturated zone pathway will include advection, linear sorption, mechanical dispersion, and molecular diffusion. Saturated zone processes and parameters represented in the PA model are described in detail in the *Saturated Zone Modeling* white paper (Appendix 7). The primary concern for the model is the breakthrough of $^{99}$Tc at the monitoring well.

#### 4.1.2.7 Air Modeling

Gaseous and particle-bound contaminants that have migrated to the surface soil layer are potentially subject to dispersion in the atmosphere. The effect of mechanical disturbance on human exposure to soil particulates is evaluated in the PA based on the effect of off-highway vehicle (OHV) use. However, although this mechanism may be consequential for human exposure, it is not likely to be a significant contributor to the overall rate of fine particulates emissions from the embankment over time. Aeolian (wind-related) disturbance is the primary cause of particulates emissions from the embankment and is the process modeled in the PA to estimate particulate emissions.

In addition to particulate emissions of contaminated surface soil due to aeolian erosion, emissions of gas-phase radionuclides diffusing across the surface of the embankment into the atmosphere are considered in the PA model. Note that this effect is counter-balanced by replacement with aeolian material that moves onto the cap. Diffusion modeling of radionuclide gases in the embankment, and estimation of flux into the atmosphere, is described in the *Unsaturated Zone* white paper (Appendix 5). For both particulate-bound and gaseous radionuclides, atmospheric dispersion modeling employing local meteorological data is conducted to calculate breathing-zone air concentrations above the embankment and at specific locations in the area where off-site receptors may be exposed (see *Dose Assessment* white paper – Appendix 11).

Atmospheric dispersion may result in significant bulk transport of fine particles modeling off of the embankment. Atmospheric dispersion modeling is also used to calculate the deposition flux of resuspended embankment particles in the areas adjacent to the embankment where ranchers and recreational receptors may be exposed. As particulates from the embankment are deposited on surrounding land, this surrounding area may become a secondary source of radionuclide exposure.

Atmospheric dispersion modeling was conducted outside of the GoldSim modeling environment, into which the model was abstracted. An atmospheric dispersion model is a mathematical model that employs meteorological and terrain elevation data, in conjunction with information on the release of contamination from a source, to calculate breathing-zone air concentrations at locations above or downwind of the release. Some models may also be used to calculate surface deposition rates of contamination at locations downwind of the release.

Both particle resuspension and atmospheric dispersion are first modeled outside of the GoldSim PA model, and the results are then incorporated into GoldSim. The particulate emission model used is a relatively simple model that has been adopted by EPA to estimate an annual-average emission rate of respirable particulates (approximately 10 μm and less, i.e., $PM_{10}$) from the ground surface. The air dispersion model used is AERMOD, which is EPA's recommended regulatory air modeling system for steady-state releases and suitable for calculating annual-average contaminant breathing zone air concentrations at various distances and in various directions from a source release. These models are described in detail in the *Atmospheric Transport Modeling* white paper (Appendix 8). Given the massive dilution that occurs for windblown sediments, it seems unlikely that this pathway will result in offsite accumulation of large amounts of transported radionuclides. Accumulation onsite seems more likely.

#### 4.1.2.8 Biological

Biological organisms play an important role in soil mixing processes, and therefore are potentially important mediators of transport of buried wastes from deeper layers to shallower layers or the soil surface. Three broad categories are evaluated for their potential effect on the redistribution of radionuclides at the Clive facility: plants, ants, and burrowing mammals. The impact of these flora and fauna will be limited largely to the top several meters, in which case, the severity of their effect on radionuclides transport might be small. Details for all three categories can be found in the *Biological Modeling* white paper (Appendix 9).

##### 4.1.2.8.1 Plants

Biotic fate and transport models have been developed to evaluate the redistribution of soils, and contaminants within the soil, by native flora and fauna. The Clive Facility is located in the eastern side of the Great Salt Lake Desert, with flora and fauna characteristic of Great Basin alkali flat and Great Basin desert shrub communities.

Plant-induced transport of contaminants is assumed to proceed by absorption of contaminants into the plants roots, followed by redistribution throughout all the tissues of the plant, both above ground and below ground. Upon senescence, the above-ground plant parts are incorporated into surface soils, and the roots are incorporated into soils at their respective depths.

Functional factors that contribute to the plant section of the biotic transport model include identifying dominant plant species, grouping plant species into categories that are significantly similar in form and function with respect to the transport processes, estimating net annual primary productivity (NAPP, a measure of combined above-ground and below-ground biomass generation), determining relative abundance of plants or plant groups, evaluating root/shoot mass ratios, and representing the density of plant roots as a function of depth below the ground surface.

Field surveys of the Clive site and surrounding areas were conducted by SWCA Environmental Consultants in September and December 2010 to identify plant species present in different vegetative associations around the Clive Site (SWCA, 2011). Five different vegetative associations were surveyed, with three associations representing the alkali flat/desert flat type soils found in the vicinity of Clive, and two associations representative of desert scrub/shrub-steppe habitat characteristic of slopes and slightly higher elevations with less-saline soil chemistry. A one hectare (100 m × 100 m) plot was established in each vegetative association, and each plot was surveyed for dominant plant species present, and the percent cover and density of each species. In addition, a small number of black greasewood, shadscale, halogeton, and Mojave seablite plants were excavated to obtain root profile measurements and above-ground plant dimensions. Plots 3 through 5 represent current vegetation at the Clive site, while Plots 1 and 2 are representative of less-saline soils that may develop on top of the waste cell cover.

A total of 41 plant species were identified on the five survey plots. Eighteen species each comprised at least 1% of the total cover on at least one plot. These 18 species were considered the most important for the purpose of modeling plant mediated transport of radiochemical contaminants at Clive. Species were grouped into five functional plant groups: grasses, forbs, greasewood, other shrubs, and trees. Greasewood is separated from other shrubs because of its status as a phreatophyte that can extend taproots in excess of five meters to reach groundwater. Annual and perennial grasses were grouped due to similar maximum rooting depths. Despite the ability of Greasewood to extend taproots, it will only do so if there is a water source to mine. There is no evidence in the Clive data that greasewood in the area of Clive extends to the water table. Also, the radon barrier acts as an impediment to deep rooting. Consequently, plant pathways for radionuclide transport are likely to have a limited effect in the current model.

#### 4.1.2.8.2 Ants

Ants fill a broad ecological niche in arid ecosystems as predators, scavengers, trophobionts and granivores. However, it is their role as burrowers that is of main concern for the purposes of this model. Ants burrow for a variety of reasons but mostly for the procurement of shelter, the rearing of young and the storage of foodstuffs. How and where ant nests are constructed plays a role in quantifying the amount and rate of subsurface soil transport to the ground surface at the Clive site. Factors relating to the physical construction of the nests, including the size, shape, and depth of the nest, are key to quantifying excavation volumes. Factors limiting the abundance and distribution of ant nests such as the abundance and distribution of plant species, and intra-specific or inter-specific competitors, also can affect excavated soil volumes. Important parameters related to ant burrowing activities include nest area, nest depth, rate of new nest additions, excavation volume, excavation rates, colony density, and colony lifespan.

Modeling soil and contaminant transport by ant species assumes that ants move materials from lower cells to those cells above while excavating chambers and tunnels within a nest. These chambers and tunnels are assumed to collapse over time and return soil from upper cells back to lower cells.

Surveys for ants at Clive were limited to surface surveys of ant colonies, including identification of ant species, measurements (length, width, and height) of ant mounds, and determination of ant nest densities in each vegetative association (SWCA, 2011). No excavations of ant nests were performed at Clive to support this initial PA model, although excavations could be conducted to support future model iterations if ant nest depth and volume are found to be sensitive parameters. Total nest depth and nest volume were extrapolated from mound surface dimensions based on correlations reported in the literature for the dominant ant species at Clive. Only two species of ants were identified during the surveys, with the western harvester ant, *Pogonomyrmex occidentalis*, accounting for 62 of the 64 nests identified. The second ant species, a member of the genus *Lasius,* was only encountered twice, both times in the mixed grassland plot. Harvester ants also tend to create the largest and deepest burrows. Consequently, the characteristics of the harvester ants were included in the model.

Although the effect of burrowing ants is modeled, it is not expected to have a large influence on model results because ant nests are not assumed to get into the waste, which is about 5m or more below ground surface for the disposal configurations considered. In addition, the design of the cap is likely to limit the potential presence of ants on the embankment. That is, the rip rap and gravel layers included in the design are not conducive to the development of ant nests.

#### 4.1.2.8.3 Burrowing Mammals

Burrowing mammals can have a profound impact on the distribution of soil and its contents near the soil surface. The degree to which mammals influence soil structure is dependent on the behavioral habits of individual species. While some species account for a large volume of soil displacement, others are less influential. Functional factors such as burrowing depth, burrow depth distributions, percent burrow by depth, tunnel cross-section dimension, tunnel lengths, soil displacement by weight, soil displacement by volume and animal density per hectare play a critical role in determining the final soil constituent mass by depth within the soil.

Modeling soil and contaminant transport by mammal species within the Clive PA model assumes animals move materials from lower cells to those cells above while excavating burrows. Burrows are assumed to collapse over time and return soil from upper cells back to lower cells. Thus, the balance of materials is preserved through time.

Each Clive plot was surveyed for small mammal burrows during September and October 2010 (SWCA 2011). Burrows were identified by animal category. Within the survey area four categories of mammal burrows were identified: ground squirrels, kangaroo rats, mouse/rats/voles, and one badger. Due to the small number of badger and ground squirrel burrows, the decision was made to treat all burrowing mammals as a single unit for modeling purposes. Small mammal trapping was conducted on the five Clive plots during the new moon in October 2010 to identify the principal small mammal fauna present in each vegetative association. Each 1-ha plot was subdivided into 25 20-m × 20-m subplots. At the center of the each subplot, two Sherman® live traps were placed, for a total of 50 traps per plot.

Deer mice (*Peromyscus maniculatus*) were the most abundant small mammal captured during trapping, and were the only mammal captured in the plots located on the Clive Facility (Plots 3, 4, and 5). Plots 3, 4, and 5 were characterized by very low mammal densities, as evidenced by both the trapping results and the burrow surveys. With such a small population in plots 3, 4, and 5, the decision was made to average these plots. It is not clear if the cap layering of the Clive embankment will be conducive to the development of mammal burrows, however, the burrows are sufficiently shallow that it is unlikely that they will have a significant impact on radionuclide transport, and hence on doses to human receptors.

#### 4.1.2.9 Erosion

The Class A South embankment is subject to erosion by the forces of wind and water. The conceptual model assumes that wind-blown material will infill the pore space between the larger materials of the cap, including the rip rap, in a short period of time. This wind-blown material has a finer particle size and moves more readily with wind or water forces acting on the cap than the rip rap or gravel. Wind blows material off-site (see Section 4.1.2.7), even while it replaces material that is removed from the cap. Water removes cap material through the formation of gullies. The large particle-sized material of the rip rap is generally considered to be resistant to movement by erosion. However, if there is sufficient disturbance by animals or OHVers, gullies are expected to form.

Once an initiating event has occurred, wherein a “nick” is formed in the rip rap of the cover (by natural or anthropogenic events), gully formation follows from water flowing in narrow channels, particularly during heavy rainfall events. Gully erosion typically results in a gully that has an approximate “V” cross section which widens (lateral growth) and deepens (vertical growth) through time until the gully stabilizes. The formation of gullies is a concern on uranium mill tailings sites and other long-term above-ground radioactive waste sites (NRC 2010). Gully erosion has the potential to move substantial quantities of both cover materials and waste, should the waste material be buried close to the surface. Gully outwash forms depositional fans on the slopes of the embankment. Gullies might form initially on the embankment through disturbance attributed to animal burrowing, or by human induced mechanisms such as cattle paths or OHV tracks.

In the Clive DU PA Model, a gully is assumed to have a triangular cross-section, with the bottom of the gully being a curved line, steeper where it initiates and flatter where the gully emerges from the embankment. The slope of the thalweg (bottom) of the gully depends on:

- the height of the gully thalweg above the mouth of the gully,
- the horizontal distance from the ridge of the embankment downslope,
- the steepness of the slope, and
- the curve of the gully thalweg, characterized by a shape parameter $b$.

Several parameters are given probability distributions to incorporate uncertainty, including *b*, the angle of repose of the gully, the angle of repose of the fan formed by the gully outwash, and the distance from the ridge at which the gully initiates. Some of these parameters may be more likely to affect whether or not a gully gets into the waste than other parameters. After parameter values are chosen for the input parameters, a system of equations is solved so that the volume of the fan (made up of the gully outwash) is the same as the volume of the gully, and so that the height of the fan is the same as the height of the gully bottom where the gully emerges from the embankment. More detail on gully calculations can be found in the *Erosion Modeling* white paper (Appendix 10).

The gully model is a simplistic model of gully erosion and landscape evolution. For example, the model assumes that 1) a gully forms instantly and doesn't change with time, 2) that between 1 and 20 gullies only are allowed to form, and 3) that gullies do not interact with other model processes such as biotic transport (e.g., no plants grow in a gully). This stylized model was used to provide a basis for discussion of whether or not gully formation is an important consideration in this waste disposal system, and to evaluate the consequences of human activities that inadvertently cause doses to future humans. To apply the effects of gully formation to doses, the average waste concentrations exposed by the gully and the average waste concentration of material removed by the gully are used. The exposure area for this waste concentration is the surface area of the fan plus the surface area of the gully for which waste layers are exposed. More detail on the dose calculations for the gully model can be found in the *Dose Assessment* white paper (Appendix 11).

#### 4.1.2.10 Dose Assessment

The dose assessment in the Model addresses potential radiation dose to any receptor who may come in contact with radioactivity released from the disposal facility into the general environment (10 CFR 61.41).The objective of a dose assessment in a radiological PA is to provide estimates of potential doses to humans over time from radioactive releases from a disposal facility after closure, as described in Section 3.3.7 of NRC (2000 – NUREG 1573). As described below, the critical groups in the Model are defined as ranchers and recreationalists.

The radiation dose limit for protection of the general population is 25 mrem/yr, as a total effective dose equivalent (TEDE). Dose limits for radiological PAs are defined in UAC Rule R313-25-19 and10 CFR 61.41as 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 of any member of the public. However, the radiation dosimetry underlying these dose metrics is based on a methodology published by the International Commission on Radiation Protection (ICRP) in 1959. More recent dose assessment methodology has been published as ICRP Publication 30 (ICRP, 1979) and ICRP Publication 56 (ICRP, 1989), employing the TEDE approach. As stated in Section 3.3.7.1.2 of NRC (2000), "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…"

The period of performance for a radiological PA defined in UAC Rule R313-25-8 requires evaluation for a minimum compliance period of 10 ky, with additional simulations for a qualitative analysis for the period where peak hypothetical dose occurs. The scope of this Model includes modeling of the disposal system performance to the time of peak hypothetical radiological dose (or peak radioactivity, as a proxy), and to quantify dose within the time frame of 10 ky.

#### 4.1.2.10.1 Receptors and Exposure Scenarios

Receptors in a PA are categorized in UAC Rule R313-25-8 and10 CFR 61.41 according to the labels "member of the public" (MOP) and "inadvertent human intruder" (IHI). The regulatory basis for, and interpretation of these categories of receptors is provided in Section 1.3. The MOP is essentially a receptor who is exposed outside the boundaries of the facility, and the IHI is someone who intrudes onto the facility and may directly contact the waste (e.g., by well drilling, or basement construction). The "critical group" receptors evaluated are modeled to receive exposure both upon the disposal embankment and in adjacent areas according to the activities foreseen (ranching and recreational uses). Both scenarios are evaluated under post-institutional control conditions. The Model may be run with or without the formation of gullies.

Ranching Scenario. The land surrounding the Clive Facility is currently utilized for cattle and sheep grazing. Ranchers typically use off-highway vehicles (OHVs, including four-wheel drive trucks) for transport. 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 pathways outlined in Table 1.

Recreational Scenario. Recreational uses on the land surrounding the Clive Facility may involve OHV use, hunting, target shooting of inanimate objects, rock-hounding, wild-horse viewing, and limited camping. As soil develops on the rip-rap surface of the cap and plant succession proceeds, the disposal unit may become more attractive for different types of recreational activities. It is assumed in the Clive DU PA Model that recreational 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, represent the most highly-exposed recreational receptors. Recreationalists may be exposed to contamination via the pathways outlined in Table 1.

**Table 1. 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 | | × (Hunter) |
| Ingestion of beef | × | |
| External irradiation – soil | × | × |
| External irradiation – immersion in air | × | × |

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. Unlike ranching and recreational receptors who might 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. Five specific off-site locations and receptors are evaluated in the Clive PA, including:

- 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 (UTTR, a military facility) to the south of the Clive facility, who may occasionally drive on an access road immediately to the west of the Clive Facility fence line;
- The resident caretaker at the east-bound Interstate-80 rest facility (the Grassy Mountain Rest Area at Aragonite) approximately 12 km to the northeast of the site, and,
- OHV riders at the Knolls OHV area (BLM land that is specifically managed for OHV recreation) 12 km to the west of the site.

#### 4.1.2.11 ALARA

CFR (Section 61.42) defines a second decision rule that pertains to populations as well as individuals. The regulation states "reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable" (or ALARA). The ALARA concept can be applied to either individuals or populations. In the context of the Clive DU PA Model, ALARA is applied to collective doses germane to the receptor populations described in the Section 4.1.2.10.

The ALARA process is also described in DOE regulations and associated guidance documents such as 10 CFR Part 834 and DOE 5400.5 ALARA (10 CFR 834; DOE 1993, 1997), and in other NRC documents (NRC, 1995, 2000). The definitions in each case are very similar; indicating that exposures should be controlled so that releases of radioactive material to the environment are as low as is reasonable taking into account social, technical, economic, practical, and public policy considerations.

The probabilistic Clive DU PA Model is designed to estimate individual annual doses to hypothetical individuals in future populations that may be exposed to radionuclide releases from the Clive Facility. The model is also able to aggregate individual doses into estimates of collective and cumulative population dose on an annual basis as well as over the 10-ky period of performance. Given this model structure, an opportunity exists with the Clive DU PA Model to evaluate ALARA in the context of population dose.

The overall implication of the various Agency regulations and guidance documents regarding ALARA is that many factors should be taken into account when considering the potential benefits of different options for disposal of radioactive waste. In order to implement ALARA in a logical system, and so that economic factors are taken into consideration, a decision analysis is implied. Decision analysis is the appropriate mechanism for evaluating and optimizing disposal, closure and long term monitoring and maintenance of a radioactive waste disposal system. Decision options for disposal at Clive include engineering options and waste placement. More generally, if decision analysis is applied, then a much wider range of options can be factored into the decision model, such as transportation of waste, risk to workers, and effect on the environment. However, for the current Model, the focus is on different options for waste disposal within the current proposed configuration of the Class A South embankment.

The decision analysis in this context is essentially a benefit-cost analysis, within which different options for the placement of waste are evaluated. For each option, the Model predicts doses to the array of receptors, and the consequences of those doses are assessed as part of an overall cost model, which also includes the costs of disposal of waste for each option. The goal is to find the best option, which is the option that provides the greatest overall benefit. The consequences of risk can be measured through a simplification that is available in ALARA guidance, including NRC 1995, which provides the basis for, and history of, assigning a dollar value to person-rem as a measure of radiation dose. Prior to the NRC (1995) guidance, a single value of $1,000 per person-rem was recommended, with the accompanying assumption that a discount rate would not be applied. The history of the selection of this value is described in NRC, 1995, and further references to prior documents. In 1995, NRC instead promoted the idea of using $2,000 per person-rem as the relevant value, subject to present worth considerations. This appears to be an overt attempt by the NRC to allow an economic decision analysis to be performed, allowing for a discount factor to be used in the assessment of ALARA. This is made clearer in NRC, 2000, which provides examples and formulas for how to implement ALARA, which include discount factors of 7% for the first 100 years, and 3% thereafter. These are steep discounting rates that result in small costs comparatively at 100 years into the future. DOE guidance also suggests that a range of $1,000 to $6,000 could be considered (DOE, 1997), but that the $2,000 value is sufficient for most purposes. The allowable range presented by DOE, however, could be used to describe uncertainty over the appropriate value.

In assigning a value to the person-rem cost to society of radiation dose, the agencies have short-circuited a full decision analysis. This is reasonable for a first pass at a decision analysis associated with the proposed disposal at Clive. Hence, the value of $2,000 is applied to the population dose. Application of the ALARA process to the Clive DU PA Model is described more completely in the *Decision Analysis* white paper (Appendix 12).

#### 4.1.2.12 Groundwater Concentrations

Apart from individual and population dose evaluations, evaluation of the PA also requires comparison of groundwater concentrations with groundwater protection levels, or GWPLs. That is, the State of Utah imposes limits on groundwater contamination, as stated in the Ground Water Quality Discharge Permit (UWQB, 2010). Part I.C.1 of the Permit specifies that GWPLs in Table 1A of the Permit shall be used for the Class A LLW Cell. Table 1A in the Permit specifies general mass and radioactivity concentrations for several constituents of interest to DU waste disposal. This includes values for mass concentration of total uranium, radium, and gross alpha and beta radioactivity concentrations for specific wells where background values were found to be in exceedence of the Table 1A limits. Part I.D.1 of the Permit specifies that the performance standard for radionuclides is 500 years. Relevant GWPLs for Clive are:

- Strontium-90 42 pCi/L,
- Technetium-99 3,790 pCi/L,
- Iodine-129 21 pCi/L,
- Thorium-230 83 pCi/L,
- Thorium-232 92 pCi/L,
- Neptunium-237 7 pCi/L,
- Uranium-233 26 pCi/L,
- Uranium-234 26 pCi/L,
- Uranium-235 27 pCi/L,
- Uranium-236 27 pCi/L, and
- Uranium-238 26 pCi/L.

The main concern for the PA model is the potential for transport of $^{99}$Tc, a contaminant in the DU waste, to the point of compliance.

Note that according to the Permit, groundwater at Clive is classified as Class IV, saline ground water, according to UAC R317-6-3 *Ground Water Classes,* and is highly unlikely to serve as a future water source. The underlying groundwater in the vicinity of the Clive site is of naturally poor quality because of its high salinity and, as a consequence, is not suitable for most human uses, and is not potable for humans. However, the Clive DU waste PA will calculate estimates of groundwater concentrations at a the location of a virtual well near the CAS embankment for comparison with the GWPLs.

#### 4.1.2.13 Deep Time Assessment

The approach to deep time modeling is briefly described in the *Conceptual Site Model for Disposal of Depleted Uranium at the Clive Facility* white paper (Appendix 2). A more in-depth discussion of the deep time modeling methodology is described in *Deep Time Assessment for the Clive PA* white paper (Appendix 13). The focus of the deep time evaluation is to assess the potential impact of glacial epoch pluvial lake events on the CAS waste embankment from 10 ky through 2.1 My post-closure. (note that this model is termed the “deep-time” model.) A pluvial lake is a consequence of periods of extensive glaciation, and results from low evaporation, increased cloud cover, increased albedo, and increased precipitation in landlocked areas. Given that long-term climatic cycles of 100 ky are considered very likely in the next 2.1 My, it is assumed that large lakes will return to the Bonneville Basin in the future. In addition to large lakes, intermediate sized lakes are also assumed to occur, periodically during a 100-ky glacial cycle. Events that might occur in deep time other than the occurrence of intermediate lakes and the cyclic return of large lakes (e.g., meteor strikes and a large eruption at Yellowstone) are not considered further in this model because their likelihood is relatively small, and their consequences are likely to be much greater and far reaching for human civilization.

For the deep time scenarios, the PA model provides a qualitative assessment of the future consequences of present-day disposal of DU waste to the environment. While no exposure or dose assessment is attempted, tracking of radioactive species concentrations provides insight into waste disposal and embankment construction design and performance. Long-term historical information on the area surrounding the Clive site is sparse, providing only a broad depiction of historical behavior of lake cycles in the Bonneville Basin. Thus, the model utilized for projecting into the long-term future is largely conceptual or stylized, providing a similarly broad depiction of future behavior

There are two components of the model used to represent the deep time scenarios. The first is modeling lake formation and dynamics in the Bonneville Basin. The second is modeling the fate of the CAS embankment and disposed DU waste.

For the first component, the deep time evaluation focuses on potential releases of radioactivity following a series of pluvial lake events caused by glacial cycles assumed to occur (approximately) every 100 ky. The 100-ky glacial periodicity is based on historical ice core and the benthic marine isotope data for the past 800 ky. These cycles are also consistent with information regarding orbital forcing, and the periodicity suggested by the Milankovitch cycles.

These 100 ky glacial cycles form the basis for modeling the return and recurrence of lake events in the Bonneville Basin. The lake formation model is applied to each 100 ky cycle similarly. One large lake is assumed to occur every in each 100 ky cycle, and several intermediate lakes are allowed to form during the transgressive and regressive phases of the large lake. Note that the current 100-ky cycle is not modeled differently than future glacial cycles, despite evidence that the current inter-glacial period might last for another 50 ky (Berger and Loutre, 2002). In the model, therefore, an intermediate lake can return sooner than might be expected in the current 100-ky cycle. The precise timing of the return of a lake at or greater than the elevation of Clive is not as important as the event itself.

For the second component, it is assumed that destruction of the CAS embankment and fate of the DU waste will result from the effects of wave action from an intermediate or large lake. In effect, it is assumed that a lake is large enough that obliteration of the embankment will occur. In this obliteration scenario, all of the embankment material above grade is dispersed across a large localized area through wave action, although this includes all the DU waste, even if some DU waste was disposed below grade. Inclusion of the below grade waste is conservative, since it allows more DU waste to migrate into returning lakes and future sediment. The waste material is mixed with sediment and then enters the lake system via dissolution. A simplifying, conservative assumption is to limit dissolution to a column above the waste dispersal area. This assumption is conservative because lake water will probably mix more extensively, creating greater dilution. As a result, these assumptions lead to greater concentrations of waste than is probably reasonable. The conservatism is included in this model because of the lack of data that exists to quantify the processes.

The deep-time model assumes that the form of DU available for deep-time transport is $U_3O_8$, which is far less soluble than $UO_3$. Fate and transport modeling performed using the PA Model indicates that the relative soluble $UO_3$ will have migrated transported to groundwater within 50 ky. Consequently, the deep time model focuses on $U_3O_8$ as the form of DU available for deep-time transport. While the lake is present, some waste in the water column will bind with carbonate ions and precipitate out into oolitic sediments, while the remaining waste will fall out with the sediment as the lake eventually recedes. The model assumes the waste is fully mixed with the accumulated sediments, a conservative assumption, since some waste is likely to be buried rather than mixed with future lake sediments. The extent of mixing of previous sediment with new sediment is not well understood; hence an assumption that the sediments completely mix is expedient, and probably leads to conservative results. All of the waste that has dissolved into the lake re-enters the lake sediment once the lake recedes. Overall sediment concentrations decrease over time because the amount of waste does not change other than through decay and ingrowth, whereas more sediment is added over time.

Thus the deep-time model should be regarded as conceptual and heuristic. The intent is to present a picture of what the long-term future might hold for the DU waste disposal embankment, rather than to provide a quantitative, temporally-specific, prediction of future conditions, or an assessment of exposure or dose to human receptors. The type of glacial climate change envisioned in the deep-time model will probably have wide-reaching consequences for the planet and human society, that are far beyond the scope of a PA for disposal of radioactive waste.

# 5.0 Model Structure

## 5.1 Summary of Important Assumptions

The results of the Clive DU PA Model depend critically on the model structure, the model specification (input probability distributions, for example) and the assumptions that underlie the model. That is, the results are fully dependent, or conditional, on the Model. The most important assumptions are identified in this section.

### 5.1.1 Points of Compliance

Points of compliance in a PA are usually defined in terms of the location in the accessible environment at which human health is evaluated in the dose assessment, and the location at which groundwater concentrations are used for comparison to GWPLs. For this model, the primary receptors (ranchers, recreators) are assumed to spend time on the site, and off the site in the general vicinity. Other receptors are defined at points in space (See Section 4.1.2.10.1). Note that the ALARA analysis addresses the same points of compliance.

Groundwater concentrations are evaluated at a virtual well located 27 m (90 ft) from the interior of the waste embankment. In the case of the proposed DU waste disposal, only the top slope section of the embankment would contain DU waste, so the effective distance from the DU waste to the well is lengthened by the width of the side slope section, to about 73 m (240 ft).

For the deep-time model, there are no receptors that are considered, and doses are not calculated. Instead, concentration of radionuclides are estimated in lake water and in lake sediment in the general vicinity of the CAS embankment.

### 5.1.2 Time Periods of Concern

There are four time periods that have import in this PA. The PA model is run fully quantitatively for dose endpoints for 10 ky. Peak mean dose is estimated and used for comparison with performance objectives for this time frame. The ALARA analysis is also performed for this period of time.

An institutional control period of 100 y is assumed, during which time doses are not calculated, because access to the site is assumed to be not possible.

Groundwater concentrations are compared to GWPLs for the first 500 years of the model, since this is the compliance period that is applied to the GWPLs under Utah Code.

The deep-time model is run for 2.1 My because the DU does not achieve secular equilibrium until about that time. That is, the model is run to peak activity of the DU, rather than to peak dose, which is undefined that far into the future.

### 5.1.3 Closure Cover Design Options

The engineered system in the PA model allows for evaluation of many different disposal configurations. DU waste is assumed to not be disposed under the side slopes. Otherwise there are 27 waste layers in the model, each about 0.5 m thick, starting with Layer 1 directly under the cap. The layers are numbered one through 27, with the 27$^{th}$ layer at the bottom of the waste cell. Layers 21 through 27 are below grade. Only one type of waste can be placed in a specific layer. Three disposal configurations are considered in this PA:

1. GDP contaminated waste in Layer 7 – SRS waste in Layer 8 – GDP uncontaminated waste in Layers 9-27. This model is termed the 3-m model, because Layer 7 is 3 m below the cap. Note that fill material is assumed for the 3 m between the cap and Layer 7.
2. GDP contaminated waste in Layer 11 – SRS waste in Layer 12 – GDP uncontaminated waste in Layers 13-27. This model is termed the 5-m model, because Layer 11 is 5 m below the cap. Note that fill material is assumed for the 5 m between the cap and Layer 11.
3. GDP contaminated waste in Layer 21 – SRS waste in Layer 22 – GDP uncontaminated waste in Layers 23-27. This model is termed the 10-m model, because Layer 21 is 10 m below the cap. Note that fill material is assumed for the 10 m between the cap and Layer 21. This model places all waste below grade.

These three configurations span a fairly wide range of options, from disposal near the cap, to disposal below grade.

### 5.1.4 Waste Concentration Averaging

Within each waste layer the contents of the waste are assumed to include the waste material and the fill material needed to occupy the layer volume. Since each layer represents a mixing cell, the concentration of the radionuclides is averaged throughout the layer. That is, each drum or cylinder is not modeled separately. This is typical of PA models, and is reasonable provided transport from the actual configuration does not differ greatly from transport from the modeled configuration.

### 5.1.5 Environmental Media Concentration Averaging

Similarly to the waste layers, concentrations in the environmental media are averaged throughout the cell that represents the medium. For example, the concentration of uranium in deep-time lake sediment is the average concentration throughout the sediment layer that is defined by its model cell.

### 5.1.6 Members of the Public

MOP is defined in terms of the receptors who perform activities in the vicinity of the Clive facility. This includes receptors at specific locations offsite as described in Section 4.1.2.10.

### 5.1.7 Inadvertent Human Intrusion

Following NRC 10 CFR 61, inadvertent intrusion is defined in terms of receptors who might perform some activities onsite. This includes ranchers, hunters and OHV enthusiasts. Inadvertent intrusion is often used in terms of direct but inadvertent access to the waste (e.g. through well drilling or basement construction), for which the initiator is exposed. However, such direct activities are unlikely at this site. The types of activities here do not result in direct exposure to the waste by the initiator, but potentially to future receptors. However, the receptors identified here are engaged in onsite activities, and are hence indirectly exposed to the DU waste.

### 5.1.8 Deep Time evaluation

The deep-time evaluation depends on the return of a lake in the Bonneville Basin that is large enough to obliterate the CAS embankment. Such a lake is assumed to occur more than once in each 100-ky glacial cycle. Once the CAS embankment is obliterated, the material is assumed to disperse within the vicinity of Clive. The dispersed waste then migrates into lake water through diffusion. All wastes that leave the sediment return to the sediment as the lake recedes, either physically or chemically. The wastes are assumed to mix with lake sediment in each lake cycle.

The outputs of interest are concentrations of radionuclides in lake water and in lake sediment.

## 5.2 Distribution Averaging

Most parameters in the Clive DU PA Model correspond to physical quantities that represent an average of some type. Some parameters represent averages over time, as they represent typical behavior that will be used throughout the 10-ky performance period, such as annual precipitation. Other parameters represent averages over space. For example, properties of vegetation represent an average vegetation effect across a model area, while soil properties represent an average across a volume of material represented by a model cell. When data are available that represent small amounts of time relative to the 10,000 years, or small areas/volumes relative to the model cells, then it is the *mean* of the data distribution that needs to be modeled.

To capture the temporal domain of the model, time steps in this type of systems-level dynamic probabilistic model are usually on the order of several to many years. Consequently, the average effects over long time frames, assuming no catastrophic changes in the system, are far more important than the effects on the scale of days, hours, minutes or seconds. Spatial and temporal scaling of available data, which are usually collected at points in time and space, is critical for the success of systems-level models. Scaling in this context is essentially an averaging process both spatially and temporally. Simple averaging works well if the effect on the response of a variable or parameter is linear. Otherwise, some care needs to be taken in the spatio-temporal averaging process. In addition, these types of models are characterized by differential equations and multiplicative terms. Averaging is a linear construct that does not translate directly in non-linear systems. Again, care needs to be taken to capture the appropriate systems-level effect when dealing with differential equations and multiplicative terms.

Another important statistical issues that is often overlooked in PA is correlation between inputs. Many parameters in the Clive DU PA Model are related to one another. One parameter may be physically constrained by the value of another parameter, or they may simply tend to vary together. When joint data are available, a simple approach is to simply calculate the sample correlation of the parameters in the data and apply the same correlation to the parameters in the model to induce a joint distribution. A simple correlation structure may not fully capture the relationship between two parameters but often provides a reasonable first approximation. Where a correlation structure is used in the Clive DU PA Model, the correlation algorithms implemented in GoldSim for Gaussian copula are used (Iman and Conover 1982, Embrechts et al. 2001).

Where data and expertise are available, it is generally preferable to construct joint distributions for the parameters by constructing a marginal distribution for one parameter and *conditional* distributions for the remaining parameters. By fitting a distinct conditional distribution of the second parameter for each possible value of the first parameter, a more realistic relationship might be constructed than can be achieved through simple correlation

The statistical methods used for appropriate spatio-temporal scaling and correlation effects are described in the *Development of Probability Distributions* white paper (Appendix 14).

## 5.3 Model Evaluation through Uncertainty and Sensitivity Analysis

The Clive DU PA is built as a probabilistic systems-level model. Systems-level modeling is geared towards decision objectives, and is a style of bottom-up modeling for which model refinement and iteration is performed in response to model evaluation. Model evaluation is performed throughout model development, but in the final stages it involves uncertainty analysis and sensitivity analysis. Quantitative assessment of the importance of inputs is necessary when the level of uncertainty in the system response exceeds the acceptable threshold specified in the decision making framework. One of the goals of sensitivity analysis is to identify which variables have distributions that exert the greatest influence on the response.

Uncertainty is captured directly for probabilistic system-level models. The input probability distributions are used to capture the range of possible parameter values. For probabilistic models, sensitivity analysis is performed simultaneously for all input parameters. This approach is termed global sensitivity analysis. It is a very powerful tool at the disposal of probabilistic modeling for identifying parameters that are important predictors of the model output, and it is not constrained by the user's preconceptions of what may be important. In addition to global sensitivity analysis, probabilistic models can be evaluated numerically in an uncertainty analysis and for value of information. Uncertainty analysis in this context involves comparison of the output distribution to performance metrics. A determination can then be made based on the comparison of the compliance of the disposal system. Value of information analysis can be performed to identify parameters for which uncertainty reduction in the output of interest might best be achieved, if it is necessary to reduce uncertainty. This approach can also be used in the context of ALARA contamination goals, to determine if further uncertainty reduction can reasonably be performed.

Sensitivity analysis is a very important tool for understanding the model. For those parameters that are deemed as important, and if the uncertainty analysis indicates, then there are options for further model refinement. These options include further data collection, and refinement of the model. Uncertainty and sensitivity analysis are applied to each endpoint (model output) separately. Consequently, it is reasonable to expect that some of the endpoints are sensitive to different inputs. For example, output doses might be sensitive to parameters that are related to radon production and transport, whereas the groundwater concentrations might be sensitive to $^{99}$Tc inventory or Kd. Consequently, each endpoint might have different needs regarding further data collection or model refinement.

Sensitivity analysis can be used to help identify those inputs for which uncertainty reduction through further information collection will have the most impact on reducing uncertainty in the model response. However, sensitivity analysis of high dimensional probabilistic models can be computationally challenging. These challenges can be met through machine learning methods applied to probabilistic simulation results. Further details are provided in the *Sensitivity Analysis Methods* white paper (Appendix 15).

Another aspect of uncertainty when running probabilistic simulations is simulation stability. The final statistics of interest might relate to the mean output, or a percentile of the output, and therefore may require a large number of simulations for stability of the estimate of the statistic. The question is, how large? The number of simulations needed can be determined by running a different number of simulations for each endpoint and statistic of interest. Otherwise, simulation uncertainty could interfere with the uncertainty and sensitivity analysis.

## 5.4 Clive DU PA Model Structure

The Clive DU PA Model is written using the GoldSim systems modeling software. Like other such models, its structure is hierarchical, with nested "Containers" providing the means to organize the model into different conceptual parts (see Figure 3). This model uses Containers to basic modeling constructs such as Materials, and contaminant transport Processes that are global (model-wide) in scope. Other containers are devoted to distinct topics, such as Inventory definitions, Disposal calculations, Exposure and Dose calculations, comparisons to GWPLs, and the development of Deep Time Scenarios. Supplemental containers define dashboards used for running the model and displaying results, collected Results from calculations around the model, Simulation Settings for model controls, and Documentation. The role of each of these is discussed below. For instructions on how to use the model, consult the *Clive DU PA Model User Guide*.

The purpose of this model is to simulate, to a degree sufficient for decision making, the fate and transport of radionuclides proposed for disposal in the Clive Facility, and to assess their potential effects on future individuals and populations. This is done in the realm of environmental transport modeling coupled with the modeling of health physics and toxicity to humans.

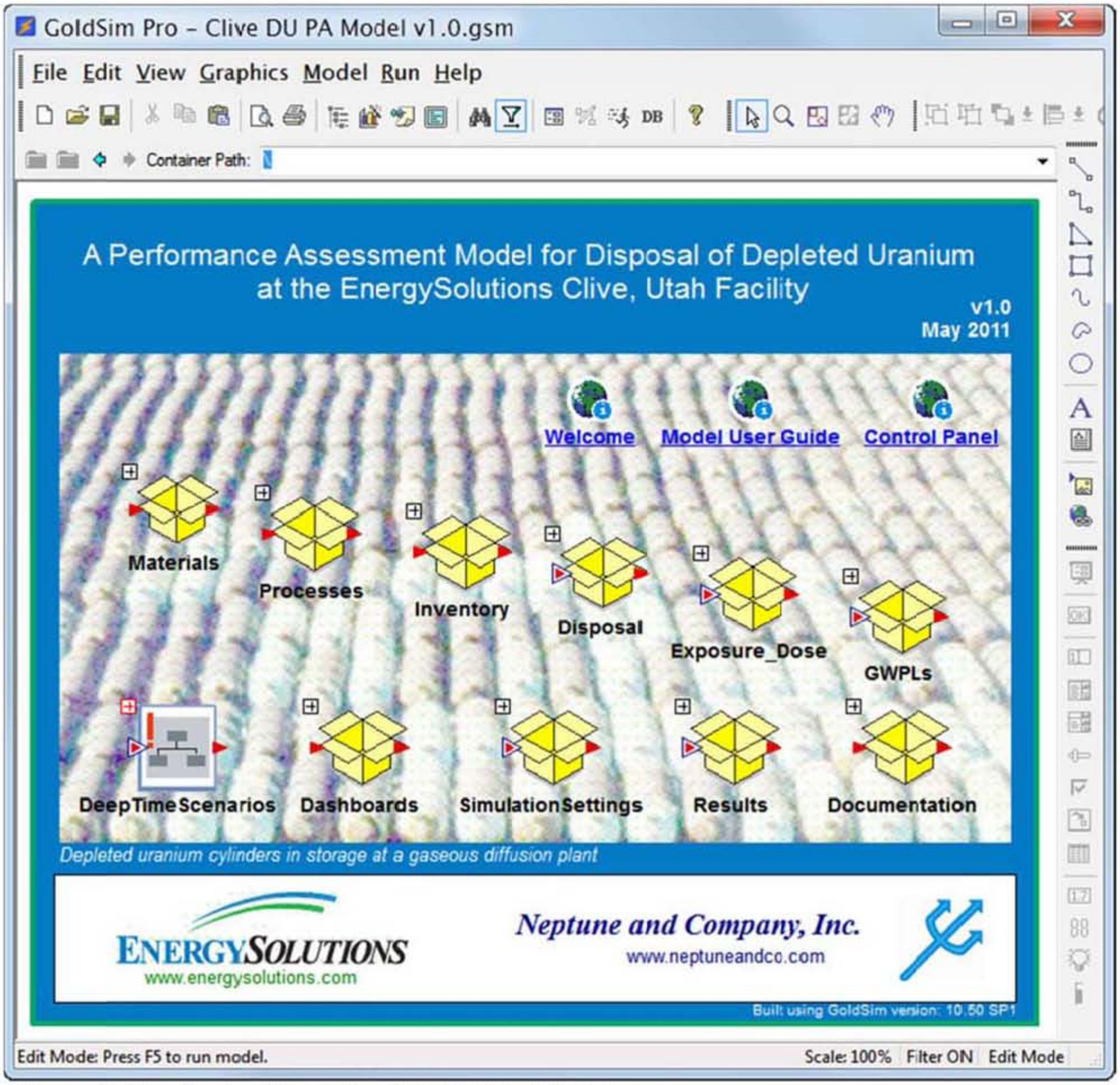


**Figure 3. Top level of the Clive DU PA Model v1.0.**

### 5.4.1 Materials

Any physical model of an environmental system must contains some sort of materials as a basis for representing the physical environment. Water, air, waste, soils, and other porous media are defined in this container, and are referenced throughout the model. The arrangement of these materials in space, and their interconnectivity, is intended to represent a large block of the environment, including the Clive Facility, or in this case the Class A South Embankment within that facility, and its surroundings. The spatial definition of the environment is in the Disposal container.

### 5.4.2 Processes

Contaminant transport in the environment is driven by several processes in this model, including advection in water, diffusion in water, diffusion in air, uptake and redistribution by plants, and disturbance by burrowing animals. These parameters defining these processes are global in model scope, and so are defined at this high level. The actual implementation of these processes in moving radionuclides in the environment, is done mostly in the Disposal container.

Radioactive decay and ingrowth, chemical solubility in water, soil/water partitioning, air/water partitioning are also fundamental processes that determine fate and transport of radionuclides, though these are defined in the Materials container, since they are directly related to materials.

### 5.4.3 Inventory

The mass of radionuclides introduced as waste into the model is called the inventory. Inside this container, the total mass of various types of DU waste is defined, as are the concentrations of the radionuclides in each type of waste. These inventories can be selected individually or in combination by the user by using the Control Panel dashboard (see Figure 4), and is then introduced to the modeling cells that represent the waste layers, in the Disposal container.

### 5.4.4 Disposal

For the first 10,000 yr following disposal, calculations are performed for the fate and transport of radionuclides from the inventory, into and throughout the modeled environment, in the Disposal container. Here the physical location of modeling cells is defined, each with materials representing what would be found at that location. For example, modeling cells representing the cover container rip rap, loess (windblown sediment), clays, and other porous media, as well as water and air. Cells representing the aquifer contain Unit 2 sediments and water, but no air, since this regions is saturated with water by definition. Waste cells contain waste and backfill as porous media, air and water, and are provided a mass of radionuclides from the inventory. As the model progresses through time, these radionuclides migrate into other part of the physical system, and eventually are found in environmental media (air, water, soils) that receptors will encounter. The Disposal container performs essentially all the contaminant transport calculations to necessary to estimate future concentrations of radionuclides in these exposure media.

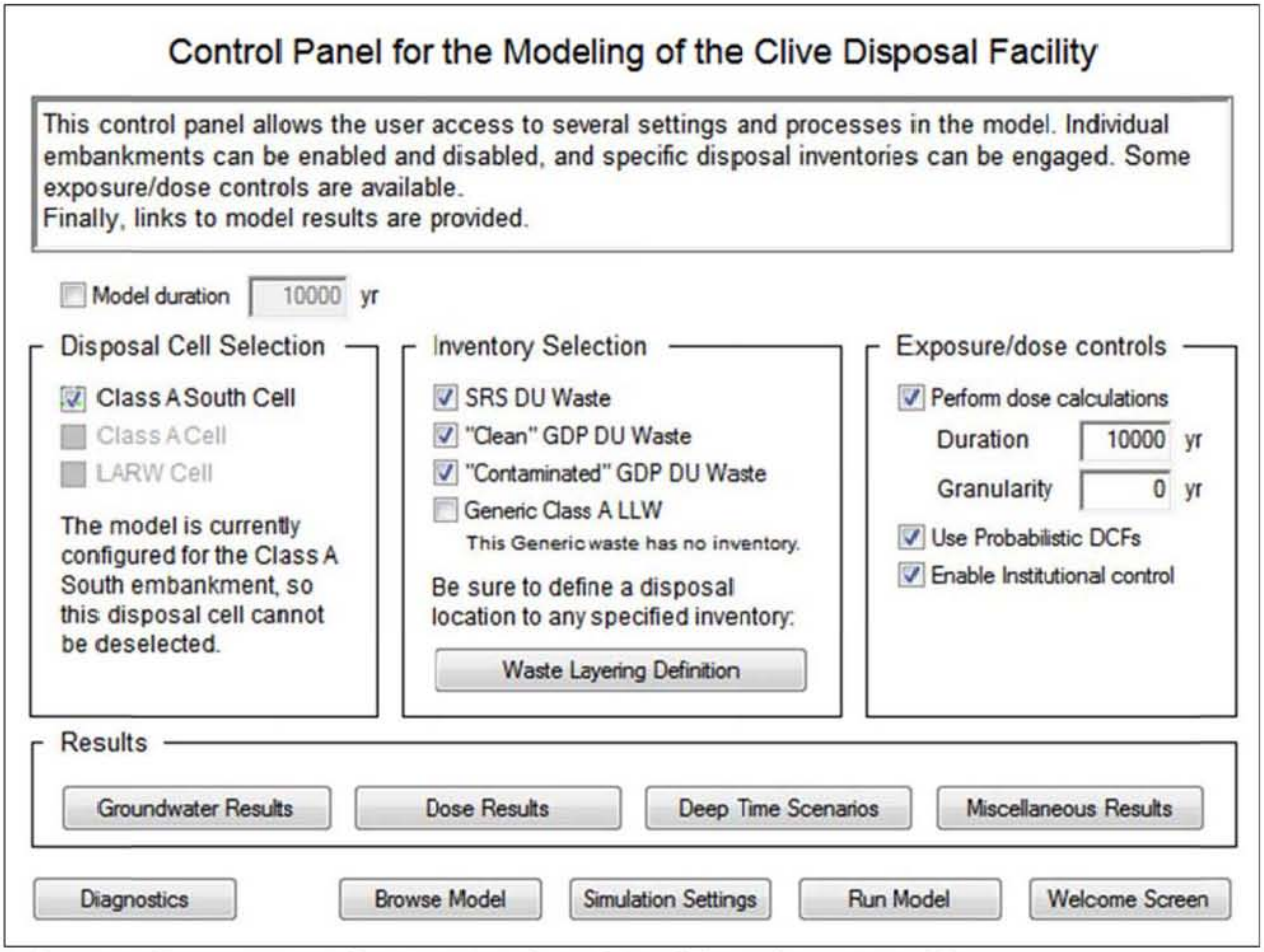


**Figure 4. Control Panel for the Modeling of the Clive Disposal Facility.**

### 5.4.5 Exposure and Dose

The exposure and dose calculations, which also include estimates of uranium toxicity hazard, are performed in this Exposure_Dose container. Receptors are hypothetical future humans who have behaviors similar to those of people around the site today: There are ranch workers, hunters, and OHV enthusiasts, all of whom are expected to have direct access to the site after institutional control is lost. There are also receptors who travel in the area, using highways, railroads, and access roads. These receptors are represented with a range of attributes and behaviors, from age to time spent on an OHV, and each encounters exposure media. As they breathe dust-laden air and walk on contaminated soils, for example, their exposures result in doses from radionuclides and toxic effects from uranium as a heavy metal. All of these calculations are performed in this container, and provide results that can be compared to performance objectives such as peak dose limits.

### 5.4.6 Groundwater Protection Level Calculations

In addition to the performance objectives provided by the State of Utah and the NRC for dose limits, there are GWPLs to be considered. In the Disposal container, the model provides radionuclide concentrations at a hypothetical monitoring well located about 27 m (90 ft) from the interior of the waste embankment. In the case of the proposed DU waste disposal, only the top slope section of the embankment would contain DU waste, so the effective distance from the DU waste to the well is lengthened by the width of the side slope section, to about 73 m (240 ft). For those radionuclides that have GWPLs defined, the maximum well concentrations within 500 yr are compared to the GWPL values. These comparison calculations are performed in the GWPLs container.

### 5.4.7 Deep Time

All the calculations described above are aimed at producing results for comparisons to performance objectives that pertain to the first 10,000 yr after disposal. Following that, and out to the time of peak activity, is considered deep time. Peak activity of the DU waste, which is predominantly $^{238}U$, is the time at which the decay products of the parent reach secular equilibrium with the parent. In this case, the peak activity is at about 2.1 million years. For the purposes of the model, then deep time is that duration from 10,000 y to 2.1 My.

Given the distinct time frame, the deep time calculations are independent of much of the rest of the model, except that the radionuclide mass in the embankment, as calculated in the Disposal container, is used as a source of radionuclides for dispersal in future lakes. The DeepTimeScenarios container produces estimates of radionuclide concentrations in the water column of future lakes, and in the sediments that they deposit.

### 5.4.8 Supplemental Containers

The Dashboards container is simply a location in the model for storing Dashboard elements, which are dialog-box-like controls for operating the model and for conveniently viewing results. The model can be executed and browsed without using any dashboards, though their convenience makes them quite useful.

The Simulation Settings container hosts a small number of elements that are used simply to control the simulation. Logical switches and values controlled by the dashboards are kept here, and the container will probably be of little interest to the average user.

The dashboards provide access to several results of general interest, most of which are collected in the Results container. In addition to those referenced by the dashboards, there are many other results that provide a more detailed look into the model. Also inside this container are the results needed for performing sensitivity analyses, such as those discussed later in this report.

Documentation contains records pertinent to model development, such as the Change Log, illustrations about particular model processes, and a large collection of references supporting the model. The subcontainer Documentation\References holds nearly 1 GB of reference materials in PDF format, and links to many more copyrighted materials that cannot be provided directly.

# 6.0 Results of Analysis

The Clive DU PA Model was run for several scenarios, in order to ascertain the effects of various assumptions. The different scenarios involved placing the DU wastes in different positions inside the waste volume in the embankment, and using the gully screening calculations. Endpoints of interest include

- groundwater concentrations of radionuclides for which GWPLs are specified,
- dose and uranium toxicity hazard to various receptors, and
- lake water and sediment concentrations of $^{238}$U in the deep time analysis.

Statistical results (e.g. mean, median, 95$^{th}$ percentile) are based on simulations of 5,000 realizations.

The waste layering scenarios include filling the embankment waste volume with the three types of DU waste, to within 3 m, 5 m, and 10 m of the bottom of the embankment cover. The top 3 to 10 m is assumed to have been backfilled with clean material. In all these cases, the waste is arranged as follows: The bottom layers, variable in number depending on the amount of clean fill used, contain Clean GDP DU, the top waste layer contains SRS DU, and the layer directly below that contains Contaminated GDP DU. Details regarding these wastes can be found in the *Waste Inventory* white paper.

Each waste layer is roughly 0.5 m (20 in) in thickness. In general, the effect of the layer is that the higher the waste is emplaced in the volume, the greater influence it has on doses, which are derived from surface soils. The lower the waste, the greater its influence on groundwater concentrations. For this reason, the contaminated DU wastes are placed above the clean DU wastes, in order to position the $^{99}$Tc that is present in contaminated wastes as far from the groundwater as possible. Details on this modeling can be found in the *Embankment Modeling* white paper. This arrangement allows exploration of a few different alternatives that help explore the PA model, and hence the performance of the system.

Groundwater protection levels are defined in the Clive Facility's groundwater discharge permit (UWQB 2009). Radionuclides with GWPLs and for which concentrations are evaluated include $^{90}$Sr, $^{99}$Tc, $^{129}$I, $^{230}$Th, $^{232}$Th, $^{237}$Np, $^{233}$U, $^{234}$U, $^{235}$U, $^{236}$U, and $^{238}$U (see Section 4.1.2.12). The Clive DU PA Model estimates contributions to groundwater concentrations from the DU wastes for 500 yr, assuming transport to a hypothetical monitoring well. Details on the groundwater transport calculations are provided in the *Unsaturated Zone Modeling* and *Saturated Zone Modeling* white papers (Appendices 5 and 7).

Possible human receptors are of the following basic types, and details are available in the *Dose Assessment* white paper (Appendix 11):

- Ranch workers (mostly ranch hands), hunters, and OHV enthusiasts are expected to be present on and near the embankment.

- Other receptors have doses evaluated at specific locations, including the nearby highway (I-80), the Knolls OHV Recreations Area (Knolls), the nearby rail road (Railroad), the Grassy Mountain Rest Area on I-80 (Rest Area), and the Utah Test and Training Range access road (UTTR).
- All receptors are considered in population dose calculations.

The formation of gullies in the embankment cap is not modeled in detail in this version of the Clive DU PA Model, but is considered rather as a screening exercise in order to assess the influence of gullies on dose and hazard calculations. The model may be run with or without consideration of this screening calculation, so that their effect may be considered, as least qualitatively. In the following presentation of results, gully screening calculations are considered in addition to the case of no gully formation. Details on the gully calculations are provided in the *Erosion Modeling* white paper (Appendix 10).

Deep time is considered to be that time after 10,000 yr, the period of performance for assessing dose as specified in the Utah regulation. Endpoints related to the deep time assessment include lake sediment concentrations of $^{238}$U, and concentrations of $^{238}$U in lakewater, when lakes are present. Details on these calculations are provided in the *Deep Time Assessment* white paper (Appendix 13).

Results for all these endpoints are presented below, summarized in tables. Graphs of time histories and of sensitivity analysis results are also shown, although in cases where results are qualitatively similar, only a single representative graph is presented.

## 6.1 Groundwater Concentrations

Peak groundwater activity concentrations within 500 yr resulting from proposed waste disposals are calculated for all radionuclides at a hypothetical monitoring well placed about 27 m (90 ft) from the interior of the waste embankment. In the case of the proposed DU waste disposal, only the top slope section of the embankment would contain DU waste, so the effective distance from the DU waste to the well is lengthened by the width of the side slope section, to about 73 m (240 ft).

### 6.1.1 Summary of Results for Groundwater

For those radionuclides for which GWPLs exist, as specified in the facility's permit (UWQB 2009), results are shown in Table 2. It should be noted that these statistics summarize the peak mean concentrations for the 500-yr period. In general, concentrations increase with time, in which case the statistics presented are the mean concentrations on or near 500 yrs. Since all modeled estimates are of mean concentrations, the statistics represent the mean, median and 95$^{th}$ percentile of the (peak of the) mean concentration. As such, the 95$^{th}$ percentile is analogous to a 95% upper confidence limit on the mean. Note that most of the distributions are markedly positively skewed, as demonstrated by the large difference between the mean and median concentrations.

**Table 2. Peak groundwater activity concentrations within 500 yr, compared to GWPLs**

| radionuclide | GWPL[1] (pCi/L) | peak activity concentration within 500 yr (pCi/L) mean | median (50th %ile) | 95th %ile |
|---|---|---|---|---|
| waste emplaced > 3 m below embankment cover | | | | |
| $^{90}$Sr | 42 | 0 | 0 | 0 |
| $^{99}$Tc | 3790 | 85.9 | 1.43e-5 | 209 |
| $^{129}$I | 21 | 0.0528 | 7.74e-21 | 0.131 |
| $^{230}$Th | 83 | 4.85e-17 | 4.19e-37 | 1.69e-26 |
| $^{232}$Th | 92 | 5.06e-23 | 0 | 1.33e-32 |
| $^{237}$Np | 7 | 1.91e-28 | 0 | 0 |
| $^{233}$U | 26 | 4.84e-13 | 5.2e-33 | 5.05e-22 |
| $^{234}$U | 26 | 2.27e-12 | 3.3e-32 | 3.3e-21 |
| $^{235}$U | 27 | 1.36e-13 | 2.7e-33 | 3.07e-22 |
| $^{236}$U | 27 | 4.41e-13 | 4.69e-33 | 4.07e-22 |
| $^{238}$U | 26 | 1.91e-11 | 2.68e-31 | 2.75e-20 |
| waste emplaced > 5 m below embankment cover | | | | |
| $^{90}$Sr | 42 | 0 | 0 | 0 |
| $^{99}$Tc | 3790 | 437 | 0.00264 | 1710 |
| $^{129}$I | 21 | 0.368 | 3.35e-16 | 1.77 |
| $^{230}$Th | 83 | 2.16e-21 | 5e-37 | 1.5e-26 |
| $^{232}$Th | 92 | 1.61e-27 | 0 | 1.28e-32 |
| $^{237}$Np | 7 | 3.93e-25 | 0 | 4.22e-38 |
| $^{233}$U | 26 | 4.43e-17 | 6.32e-33 | 3.85e-22 |
| $^{234}$U | 26 | 2.66e-16 | 3.65e-32 | 2.44e-21 |
| $^{235}$U | 27 | 2.93e-17 | 2.99e-33 | 2.08e-22 |
| $^{236}$U | 27 | 3.61e-17 | 5.16e-33 | 3.35e-22 |
| $^{238}$U | 26 | 2.23e-15 | 2.98e-31 | 1.97e-20 |
| waste emplaced > 10 m below embankment cover | | | | |
| $^{90}$Sr | 42 | 0 | 0 | 0 |
| $^{99}$Tc | 3790 | 14400 | 113 | 81400 |
| $^{129}$I | 21 | 12.7 | 5.8e-07 | 81.2 |
| $^{230}$Th | 83 | 1.53e-21 | 3.81e-37 | 1.23e-26 |
| $^{232}$Th | 92 | 1.34e-27 | 0 | 9.28e-33 |
| $^{237}$Np | 7 | 7.6e-18 | 0 | 4.69e-26 |
| $^{233}$U | 26 | 2.92e-17 | 2.25e-32 | 4.67e-22 |
| $^{234}$U | 26 | 1.57e-16 | 2.99e-32 | 2.14e-21 |

| | | | | |
|---|---|---|---|---|
| $^{235}U$ | 27 | 1.61e-17 | 2.64e-33 | 1.82e-22 |
| $^{236}U$ | 27 | 2.4e-17 | 4.28e-33 | 3.23e-22 |
| $^{238}U$ | 26 | 1.4e-15 | 2.41e-31 | 1.68e-20 |

[1]GWPLs are from UWQB (2009) Table 1A.

Based on these results, it can be seen that as the waste is emplaced lower in the embankment, monitoring well concentrations increase. This makes sense for two reasons: 1) The waste is closer to the groundwater, and so has a shorter travel distance, bringing the peak closer in time, and 2) the waste is more concentrated since it is arranged into a smaller volume, thereby decreasing the duration of breakthrough at the well, and increasing its amplitude.

For most radionuclides in Table 2 the groundwater concentrations are negligible compared to the GWPLs. The exceptions are $^{99}$Tc and $^{129}$I. For the 10-m model, the 95$^{th}$ percentiles for these two radionuclides exceed their GWPLs (this is also the case for the mean for $^{99}$Tc). However, the median is still much less than the respective GWPLs. The distributions of these concentrations are very skewed, largely because of the skew in some if the input distributions. For example, the distributions for $K_d$ are expressed as log-uniform.

In the case of $^{129}$I, this radionuclide was not detected in any samples collected from the SRS drums (see the *Waste Inventory* white paper – Appendix 4). Not only was $^{129}$I not detected, but it was not identified in any sample. However, the detection limits were used directly for creating the input distribution for inventory of $^{129}$I. Consequently, the results presented are based on data that suggest that $^{129}$I does not exist in the SRS inventory.

In the case of $^{99}$Tc there are concerns over both the inventory concentration distribution, the concentration of $^{99}$Tc in the GDP waste, and the infiltration modeling. The $^{99}$Tc inventory concentration distribution is derived from three datasets that suggest very different concentrations. Consequently, the input distribution covers more than one order of magnitude of possible $^{99}$Tc concentrations. With more data or better information, it is reasonable to expect that this uncertainty could be reduced. In addition, the use of HELP for infiltration modeling in an arid setting is known to over-estimate infiltration rates (see Section 4.1.2.4). The model results suggest that groundwater concentrations of $^{99}$Tc are less than the GWPL for the 3-m and 5-m configurations, but that some of the simulations exceed this threshold in the 10-m model. With some model refinements that address the inventory distributions and the infiltration rates, the results are more likely to be reduced.

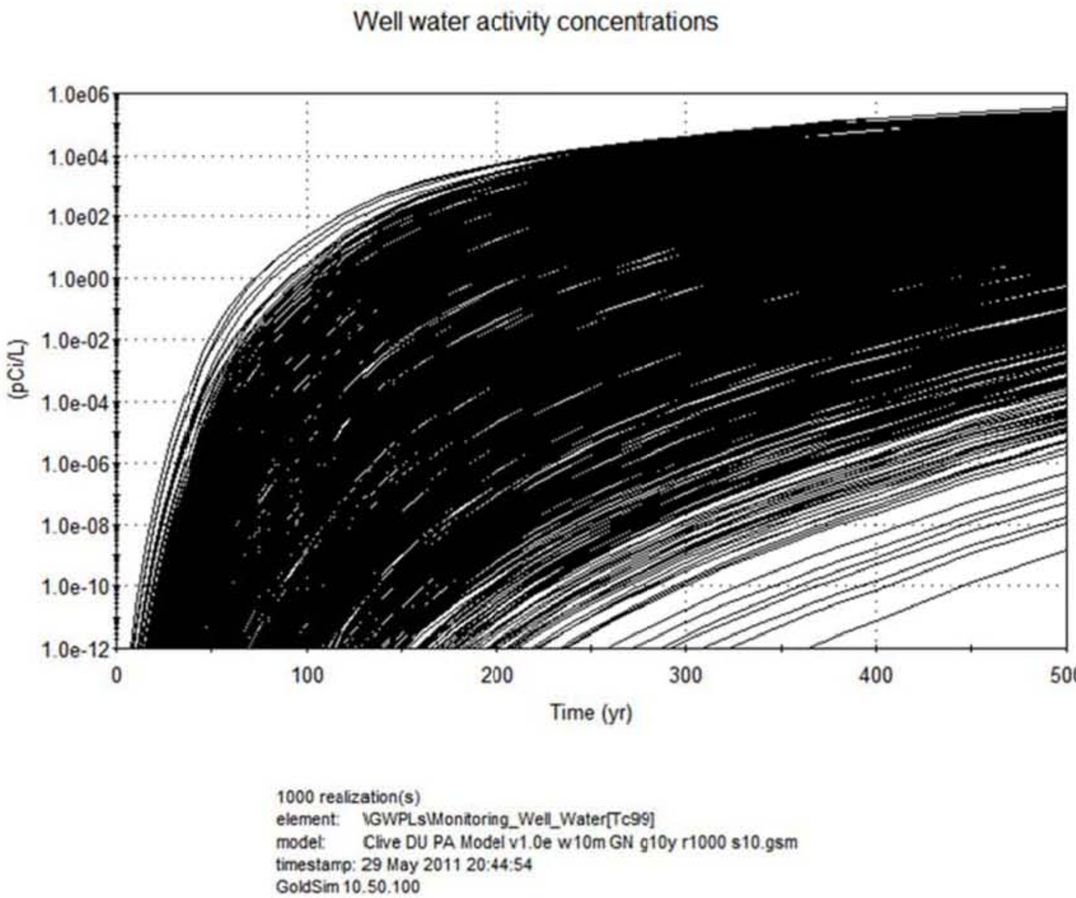


**Figure 5. Time history of mean peak $^{99}$Tc well concentrations: all realizations.**

Technetium-99 is selected to represent a time history of monitoring well concentrations, as shown in Figure 5. This time history is for the case where waste is emplaced greater than 10 m below the embankment cover, and therefore represents the highest concentrations of the three waste layering cases. Figure 5 shows each of the 1,000 realizations, and Figure 6 shows a statistical summary of those realizations. For clarity of presentation, these graphs show a suite of 1,000 realizations rather than the full 5,000 realizations on which the summary statistics in Table 2 are based. Subsequent time histories will show only the statistical summaries. Of particular interest is the increase in concentrations of $^{99}$Tc over time up to the 500-yr compliance period.

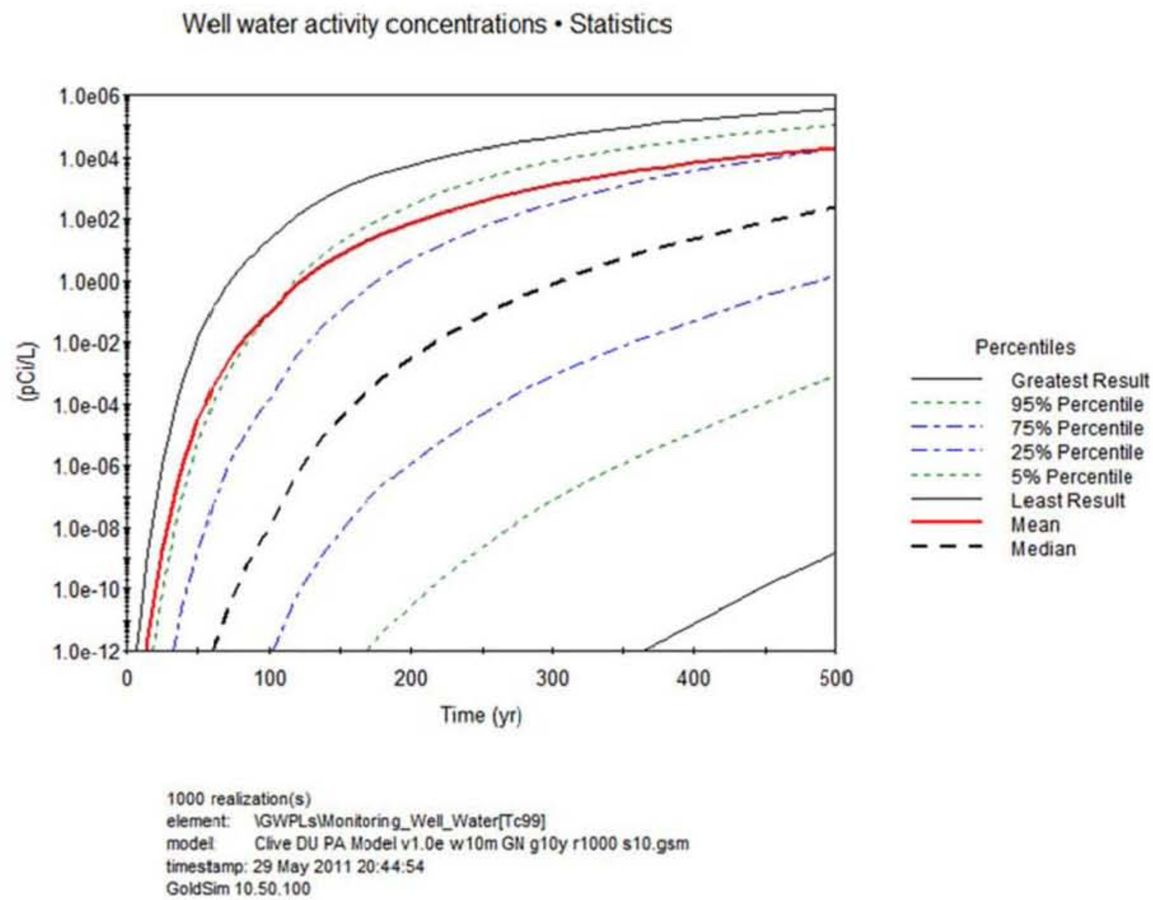


**Figure 6. Time history of mean peak $^{99}$Tc well concentrations: statistical summary.**

### 6.1.2 Sensitivity Analysis for Groundwater

A sensitivity analysis of the $^{99}$Tc groundwater concentrations was performed in order to determine which modeling parameters are most significant in predicting its value. As seen in Figure 7, the concentration is sensitive to the value of the soil/water partition coefficient, $K_d$, for Tc in the Unit 3 sand, which is used to represent the waste as well as the unsaturated zone below the embankment. In this case, which is for waste emplaced 3 m below the cover, the sand $K_d$ for Tc accounts for 83.4% of the variation. No other model input parameters account for more than 5%, and so are not shown. In all other cases of $^{99}$Tc groundwater concentrations, a similarly strong and almost exclusive dependence on sand $K_d$ is seen.

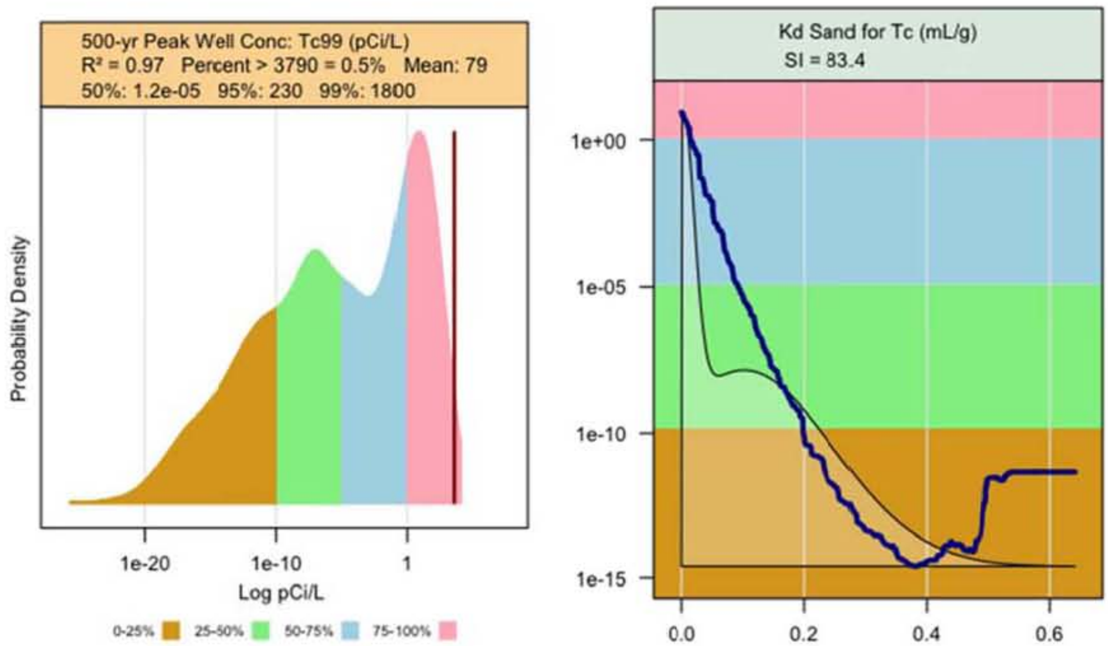


**Figure 7. Partial dependence plot for peak $^{99}$Tc groundwater concentration, assuming waste at 3 m.**

Note that the input distribution is bimodal. This is because the distribution assumes a non-negligible probability that the $K_d$ is zero, and a complementary probability that the $K_d$ is greater than zero. The partial dependence plot looks reasonable for the range of values of $K_d$ up to about 0.4. After that, there are too few data simulated from the input distributions to provide a good fit, however, the response is very small concentrations.

## 6.2 Receptor Doses

Doses to receptors are calculated as total effective dose equivalent (TEDE), and are to be compared to the performance objective of a peak dose of 0.25 mSv (25 mrem) in a year, achieved within 10,000 yr (Utah 2010). Comparison with the inadvertent intrusion standard of 5 mSv (500 mrem) in a year can also be considered for the models that include human induced gully erosion.

### 6.2.1 Summary of Results for Doses

These are summarized in two tables: Table 3 shows the statistics for mean TEDE for all receptors, without the gully screening calculations, for the cases of waste emplaced at 3 m, 5 m, and 10 m below the embankment cover.

**Table 3. Peak mean TEDE, without consideration of gullies: statistical summary**

| | Peak TEDE (mrem in a yr) within 10,000 yr | | |
|---|---|---|---|
| **receptor** | **mean** | **median (50$^{th}$ %ile)** | **95$^{th}$ %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 4.37 | 3.44 | 11.3 |
| hunter | 0.187 | 0.152 | 0.462 |
| OHV enthusiast | 0.286 | 0.234 | 0.721 |
| I-80 receptor | 0.000124 | 9.85e-5 | 0.000318 |
| Knolls receptor | 0.00129 | 0.000989 | 0.00336 |
| rail road receptor | 0.000194 | 0.000155 | 0.0005 |
| rest area receptor | 0.00249 | 0.002 | 0.00633 |
| UTTR access road receptor | 0.0617 | 0.0493 | 0.156 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.598 | 0.473 | 1.52 |
| hunter | 0.0258 | 0.021 | 0.0628 |
| OHV enthusiast | 0.039 | 0.0321 | 0.0947 |
| I-80 receptor | 1.44e-5 | 1.20e-5 | 3.54e-5 |
| Knolls receptor | 0.000147 | 0.000117 | 0.000383 |
| rail road receptor | 2.26e-5 | 1.89e-5 | 5.56e-5 |
| rest area receptor | 0.000289 | 0.000245 | 0.00073 |
| UTTR access road receptor | 0.0071 | 0.00589 | 0.0177 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00596 | 0.00471 | 0.0152 |
| hunter | 0.000253 | 0.000205 | 0.000624 |

| | | | |
|---|---|---|---|
| OHV enthusiast | 0.000388 | 0.000313 | 0.00094 |
| I-80 receptor | 1.54e-7 | 1.21e-7 | 3.89e-7 |
| Knolls receptor | 1.62e-6 | 1.22e-6 | 4.33e-6 |
| rail road receptor | 2.42e-7 | 1.9e-7 | 6.11e-7 |
| rest area receptor | 3.13e-5 | 2.51e-6 | 7.84e-6 |
| UTTR access road receptor | 7.81e-5 | 6.16e-5 | 0.0002 |

Table 4 shows the same information, but with the gully screening calculations included. That is, these doses evaluate the effects of the formation of a small number of gullies on the TEDE for all receptors.

Note that the doses to the offsite receptors are very small. Consequently, these receptors are not considered further. Of greater interest are the doses to the ranchers, hunters and OHVers. These three classes of receptors were modeled with the intent of capturing dose to each hypothetical individual in the relevant populations (see the *Dose Assesssment* white paper – Appendix 11). The data presented hence represent summary statistics for the peak average dose to each group of receptors. The peak of the average doses is a reasonable surrogate for average doses at 10,000 years in this PA model, because dose increases with time for DU. Consequently, the 95$^{th}$ percentile is analogous to a 95% upper confidence limit of the mean dose that is typically used under CERCLA, for example.

The output dose distributions are very positively skewed, with long tails. The long tails are probably due to a combination of factors that include: skewed input distributions that reasonably reflect uncertainty in upper values of a parameter; multiplicative effects in the model; and, missing correlations between some input parameters, which can lead to implausible combinations of input values. Consequently, dose results that are far into the tail of the output dose distributions might be unreliable. The mean and 95$^{th}$ percentile are used for comparison with performance objectives.

The greatest doses are shown for the 3-m configuration, as would be expected. The doses to ranch workers are greater than to the other receptors. However, in all cases the summary statistics present values that are less than the MOP performance objective of 0.25 mSv.

When gullies are included in the model, in the stylized fashion in which they are modeled, the doses are increased. This is because of both thinning of the cover layers (cap and fill material), and possible direct exposure to the DU waste. Given the flexibility available in the configuration options in the model, other options could also be considered than the three configurations presented here.

**Table 4. Peak mean TEDE, with gully screening calculation: statistical summary**

| receptor | Peak TEDE (mrem in a yr) within 10,000 yr | | |
|---|---|---|---|
| | mean | median (50th %ile) | 95th %ile |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 20.9 | 11.6 | 72.3 |
| hunter | 0.8 | 0.467 | 2.62 |
| OHV enthusiast | 1.22 | 0.729 | 3.99 |
| I-80 receptor | 0.000123 | 0.0001 | 0.000315 |
| Knolls receptor | 0.0013 | 0.001 | 0.00341 |
| rail road receptor | 0.000193 | 0.000157 | 0.000495 |
| rest area receptor | 0.00247 | 0.00202 | 0.00621 |
| UTTR access road receptor | 0.061 | 0.0499 | 0.156 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.564 | 0.443 | 1.44 |
| hunter | 0.0244 | 0.0198 | 0.0603 |
| OHV enthusiast | 0.0367 | 0.0298 | 0.0898 |
| I-80 receptor | 1.47e-5 | 1.18e-5 | 3.76e-5 |
| Knolls receptor | 0.000154 | 0.00012 | 0.000405 |
| rail road receptor | 2.32e-5 | 1.86e-5 | 5.90e-5 |
| rest area receptor | 0.000298 | 0.000241 | 0.000746 |
| UTTR access road receptor | 0.00732 | 0.00587 | 0.0185 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00594 | 0.00457 | 0.0155 |
| hunter | 0.000257 | 0.000203 | 0.000636 |
| OHV enthusiast | 0.000386 | 0.000306 | 0.00096 |
| I-80 receptor | 1.58e-7 | 1.25e-7 | 3.95e-7 |
| Knolls receptor | 1.64e-6 | 1.24e-6 | 4.35e-6 |
| rail road receptor | 2.48e-7 | 1.97e-7 | 6.2e-7 |
| rest area receptor | 3.17e-6 | 2.49e-6 | 7.86e-6 |
| UTTR access road receptor | 7.83e-5 | 6.18e-5 | 0.000199 |

### 6.2.2 Sensitivity Analysis for Doses

Sensitivity analysis was performed on the results for the mean TEDE to ranch workers, hunters, and to OHV enthusiasts. The partial dependence plots for the ranch worker, assuming waste emplacement at 3 m and no gullies, is shown in Figure 8. In this case, the radon E/P ratio (escape-to-production ratio) is the most significant predictor of dose, followed by the moisture content in the sacrificial soil layer, and the $K_d$ for radium in sandy soils. All other cases of the sensitivity analysis for dose are qualitatively similar. It is not surprising that all these receptors have such similar influences from the same parameters, since they engage in similar behaviors.

The sensitive parameters are all associated with the impact of radon on the doses. Radium is the pre-cursor to radon in the decay chain; increased moisture content mitigates radon transport; and, the radon E/P ratio affects the amount of radon that can leave the system. Radon is the greatest dose driver in the model. Of interest is the approach to estimating radon dose, which is documented in the *Dose Assessment* white paper (Appendix 11). Radon dose is not often calculated in a PA. Instead, radon flux at the surface of a disposal system is calculated. This example perhaps indicates the important of radon in a dose calculation, although it also indicates that the predicted doses are very low without the contribution from radon.

Similar sensitivity results are found for the no gully models, and for the gully models in the 5-m and 10-m cases. In these cases the gullies have limited effect because the depth of the gullies does not penetrate the waste.

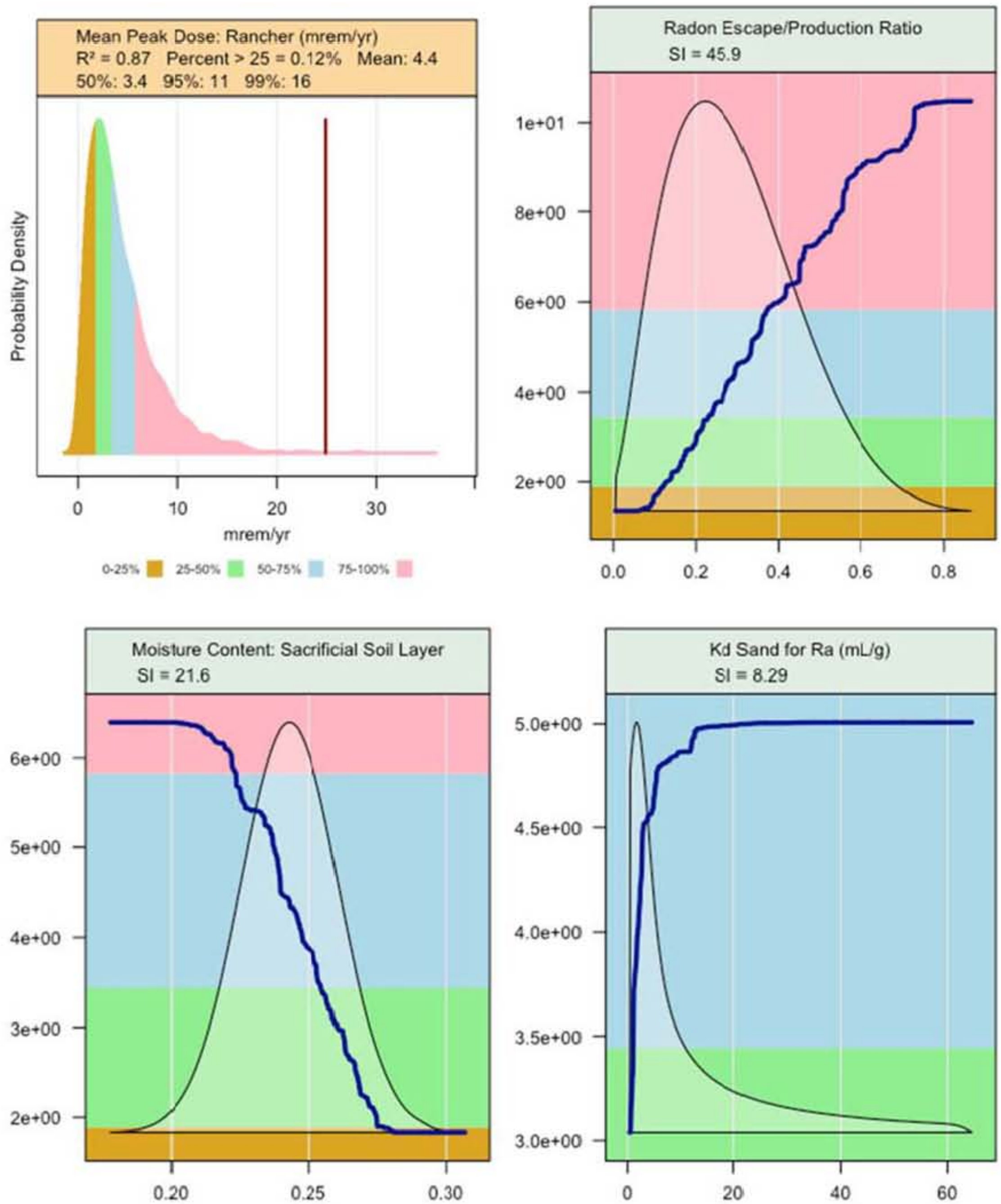


**Figure 8. Partial dependence plots for the mean ranch worker dose, assuming waste at 3 m and no gullies**

However, very different sensitivities are found for the gully screening calculations with waste emplaced up to 3 m below the embankment cover. As seen in Figure 9, the sensitive parameters are all gully-related: Angle of repose of the debris fan is the most significant, describing about half of the variation.

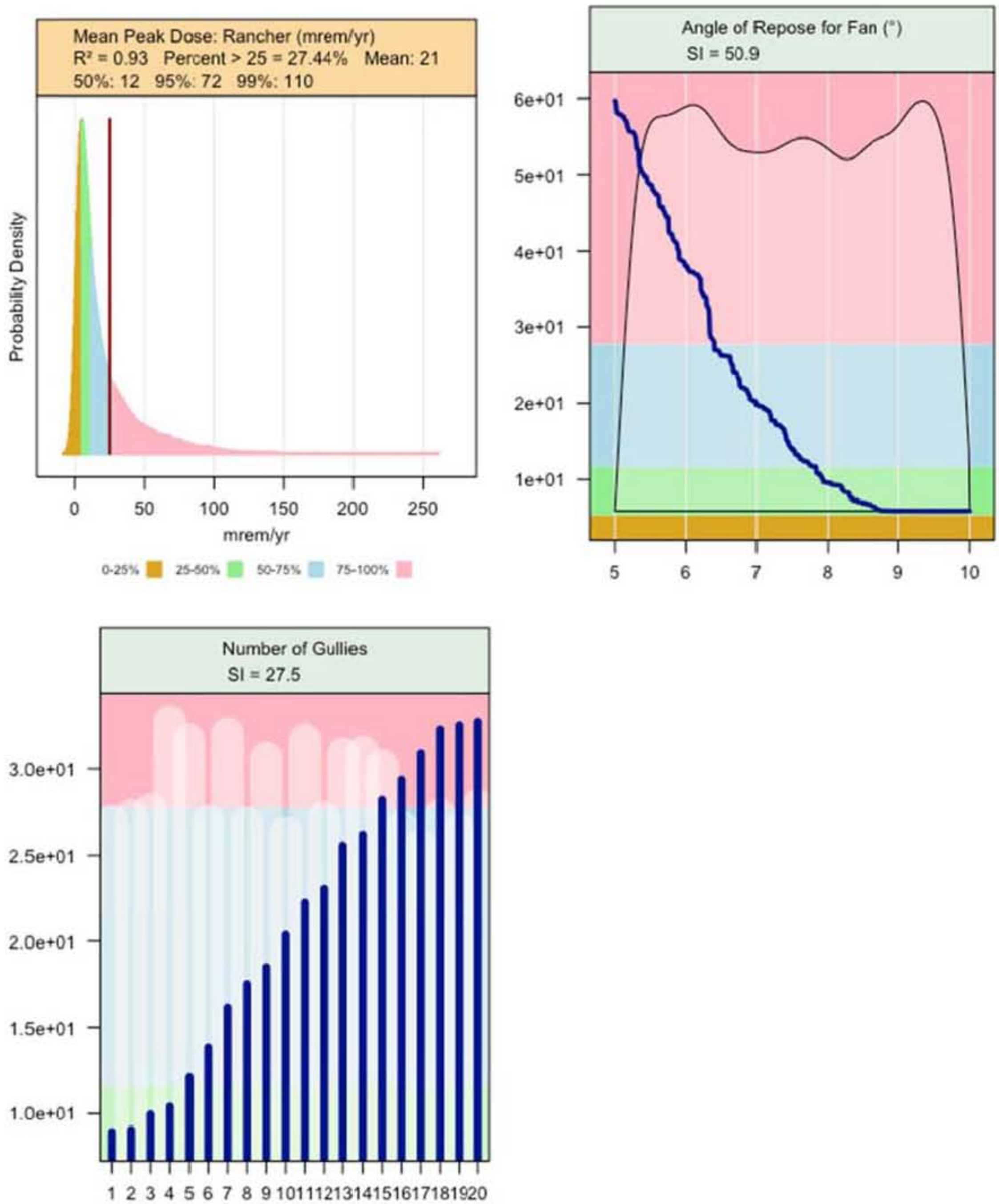


**Figure 9. Partial dependence plots for the mean ranch worker dose, assuming waste at 3 m, with gullies**

This is followed by the number of gullies, which was defined as a small number (1 to 20) just to see if they were significant. It is clear from these results that, if the waste is buried within 3 m of the cover, gullies are quite significant in contributing to doses. If the waste is emplaced at 5 or 10 m below the embankment cover, this sensitivity to gully formation goes away. A summary of sensitive parameters for each endpoint is provided in the following tables, showing for each of the principal receptors (ranch worker, hunter, and OHV enthusiast) the sensitivity to input parameters for the various waste emplacement depth cases, and with (Table 5) or without (Table 6) gullies. Only those input parameters with a sensitivity index (SI) over 5% are shown.

**Table 5. Sensitivities of peak mean TEDE within 10,000 yr, with gully screening calculation**

| receptor | SI rank | input parameter | sensitivity index (SI) |
|---|---|---|---|
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 1 | Angle of repose of the outwash fan | 51 |
| | 2 | Number of gullies | 28 |
| hunter | 1 | Angle of repose of the outwash fan | 52 |
| | 2 | Number of gullies | 29 |
| OHV enthusiast | 1 | Angle of repose of the outwash fan | 54 |
| | 2 | Number of gullies | 29 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 1 | Radon escape/production ratio | 46 |
| | 2 | Sacrificial soil water content | 16 |
| hunter | 1 | Radon escape/production ratio | 49 |
| | 2 | Sacrificial soil water content | 18 |
| | 3 | Radium $K_d$ in sand | 7.7 |
| OHV enthusiast | 1 | Radon escape/production ratio | 50 |
| | 2 | Sacrificial soil water content | 18 |
| | 3 | Radium $K_d$ in sand | 7.7 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 1 | Radon escape/production ratio | 35 |
| | 2 | Sacrificial soil water content | 14 |
| hunter | 1 | Radon escape/production ratio | 26 |
| | 2 | Sacrificial soil water content | 9.6 |
| OHV enthusiast | 1 | Radon escape/production ratio | 30 |
| | 2 | Sacrificial soil water content | 11 |

**Table 6. Sensitivities of peak mean TEDE within 10,000 yr, with no gullies**

| receptor | SI rank | input parameter | sensitivity index (SI) |
|---|---|---|---|
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 1 | Radon escape/production ratio | 46 |
| | 2 | Sacrificial soil water content | 22 |
| | 3 | Radium $K_d$ in sand | 8.3 |
| hunter | 1 | Radon escape/production ratio | 50 |
| | 2 | Sacrificial soil water content | 24 |
| | 3 | Radium $K_d$ in sand | 9.0 |
| OHV enthusiast | 1 | Radon escape/production ratio | 51 |
| | 2 | Sacrificial soil water content | 24 |
| | 3 | Radium $K_d$ in sand | 9.1 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 1 | Radon escape/production ratio | 42 |
| | 2 | Sacrificial soil water content | 19 |
| | 3 | Radium $K_d$ in sand | 8.4 |
| hunter | 1 | Radon escape/production ratio | 47 |
| | 2 | Sacrificial soil water content | 23 |
| | 3 | Radium $K_d$ in sand | 9.5 |
| OHV enthusiast | 1 | Radon escape/production ratio | 49 |
| | 2 | Sacrificial soil water content | 23 |
| | 3 | Radium $K_d$ in sand | 9.9 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 1 | Radon escape/production ratio | 37 |
| | 2 | Sacrificial soil water content | 18 |
| hunter | 1 | Radon escape/production ratio | 38 |
| | 2 | Sacrificial soil water content | 18 |
| | 3 | Radium $K_d$ in sand | 8.7 |
| OHV enthusiast | 1 | Radon escape/production ratio | 37 |
| | 2 | Sacrificial soil water content | 17 |
| | 3 | Radium $K_d$ in sand | 8.6 |

In the cases where gullies do not form, or they do not encounter waste, receptor doses are sensitive to three parameters related to radon, implying that the dose from radon is important. The most sensitive is the radon E/P ratio, which defines the fraction of $^{222}$Rn that escapes into the mobile environment, when formed by radioactive decay from its parent, $^{226}$Ra. Radon that does not escape, such as that trapped in a crystalline matrix, stays in place and decays to polonium and then to $^{210}$Pb. Note that the higher the E/P ratio, the higher the dose.

The significance of the water content in the sacrificial soil layer is that as radon diffuses upward through the engineered cover system, whatever small amount gets through the radon barrier then migrates up through the sacrificial soil. Since radon has a propensity to partition from air into water (a high Henry's Law constant), the wetter a porous medium is, the slower radon will migrate through it. In this case, the higher the moisture content, the lower the dose.

The third most significant parameter in these cases is the soil/water partition coefficient ($K_d$) in sand for radium. The link to dose is most likely through radon, given the other two parameters. In the Clive DU PA Model, the DU wastes are assigned $K_d$ values assuming the geochemistry of their constituents is dominated by the fill materials surrounding the waste. This fill material is derived from the Unit 3 stratum, which is a sandy soil assigned $K_d$ for sand. The sensitivity plot (Figure 8) shows that as the Ra $K_d$ increases, especially at low values, the dose increases. A high $K_d$ value would tend to make the radium partition onto soils, rather than migrate with local infiltrating water. If $^{226}$Ra is not leaving the waste layers due to a high $K_d$, then it remains behind as a source for $^{222}$Rn. A plausible transport and exposure scenario is that $^{222}$Rn is formed by the decay of $^{226}$Ra, the radon diffuses upward to the ground surface, partitioning into pore water along the way, and there enters the atmosphere where it is dispersed and inhaled by receptors. A combination of high radium $K_d$ in sand, low water content in sacrificial soil, and high radon E/P ratio makes for high doses, and *vice versa*.

Also of significance is the formation of gullies in the 3-m model. It is no surprise that in the presence of gullies, the parameters defining gully formation are the most important. The angle of repose of the outwash fan is part of what defines the area of that fan (which is much larger than the area of the narrow but deep gully), and the number of fans multiplies that area directly. Since these receptors spend time on the fans as a proportion of the total fan area compared to the entire embankment area, the amount of time spent on the fan (and in the gullies themselves), where wastes may be exposed, is important in determining the doses. If wastes are not exposed, as in the cases of waste emplacement greater than 5 or 10 m below the embankment cover, the gullies do not generally encounter wastes, and therefore the outwash is not contaminated, and there is no associated contribution to dose. This analysis strongly indicates that the intrusion of gullies into waste-bearing layers nearer the top of the embankment is what is important.

## 6.3 Receptor Uranium Hazard Quotients

Uranium hazard quotients (HQs) to receptors within 10,000 yr are calculated, and are compared to EPA's standard point of departure for hazard index of 1. Uranium hazard is not regulated for disposal of radioactive waste. However, it provides another point of reference for evaluating site performance for the disposal of DU.

### 6.3.1 Summary of Results for Uranium Hazard

The uranium hazard results are summarized in two tables: Table 7 shows the statistics for mean uranium hazard quotient for all receptors, without the gully screening calculations, for the cases of waste emplaced at 3 m, 5 m, and 10 m below the embankment cover.

Table 8 shows the same information, but with the gully screening calculations included. That is, these results evaluate the effects of the formation of a small number of gullies on the uranium hazard quotient for all receptors. In the case of no gullies the hazard quotients for uranium are extremely small, indicating essentially no risk from uranium toxicity. The results in Table 8 are similar for both the 5-m and 10-m waste configurations. However, the 3-m configuration shows some simulations with comparatively large uranium hazard quotients. For example, the 95$^{th}$ percentile is 47.8 for ranchers. Several other values are greater than 1. Similar to the dose results presented above, this indicates that disposal of DU waste near to the top of the embankment is not as protective of human health and the environment. However, disposal of DU waste below 5 m appears to demonstrate clear compliance with the performance objectives.

**Table 7. Peak mean uranium hazard quotient, without consideration of gullies: statistical summary**

| | Peak uranium hazard quotient within 10,000 yr | | |
|---|---|---|---|
| **receptor** | **mean** | **median (50$^{th}$ %ile)** | **95$^{th}$ %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 4.81e-7 | 1.15e-10 | 5.24e-7 |
| hunter | 1.15e-8 | 3.38e-12 | 1.32e-8 |
| OHV enthusiast | 1.07e-8 | 1.31e-12 | 7.53e-9 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 5.35e-8 | 9.64e-14 | 8.51e-9 |
| hunter | 1.34e-9 | 2.65e-15 | 2.57e-10 |
| OHV enthusiast | 1.27e-9 | 9.6e-16 | 1.66e-10 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 1.23e-10 | 3.23e-21 | 7.83e-12 |
| hunter | 3.44e-12 | 9.62e-23 | 2.42e-13 |
| OHV enthusiast | 2.37e-12 | 3.86e-23 | 1.1e-13 |

**Table 8. Peak mean uranium hazard quotient, with gully screening calculation: statistical summary**

| | Peak uranium hazard quotient within 10,000 yr | | |
|---|---|---|---|
| **receptor** | **mean** | **median (50th %ile)** | **95th %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 8.62 | 1.47 | 47.8 |
| hunter | 0.199 | 0.0331 | 1.11 |
| OHV enthusiast | 0.238 | 0.0394 | 1.34 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.00566 | 7.02e-6 | 0.024 |
| hunter | 0.000129 | 1.62e-7 | 0.000514 |
| OHV enthusiast | 0.000155 | 1.97e-7 | 0.000642 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 5.15e-6 | 1.05e-12 | 8.29e-7 |
| hunter | 1.55e-7 | 2.75e-14 | 1.92e-8 |
| OHV enthusiast | 2.22e-7 | 3.43e-14 | 2.46e-8 |

### 6.3.2 Sensitivity Analysis for Uranium Hazard Quotient

Sensitivity analysis was performed on the results for the mean uranium hazard quotient to ranch workers, hunters, and to OHV enthusiasts. The partial dependence plots for the ranch worker, assuming waste emplacement at 3 m and no gullies, is shown in Figure 10. In this case, the water content exponent for the water tortuosity model is the most significant predictor of the hazard quotient, followed by the molecular diffusivity in water, the porosity exponent for the water tortuosity model, the solubility of $UO_3$, and the vegetation association selector. The graphs in Figure 10 are specific to the ranch worker receptor, but as can be seen from consulting the values in Table 9, this is representative to all three major receptor types, in the case of waste emplacement at 3 m below the embankment cover, and no consideration of gullies.

Two of these are related to the water tortuosity model, where tortuosity in the water phase equals the water content (to a power "water content exponent") divided by porosity (to the power "porosity exponent"). Therefore, higher values of "water content exponent" cause higher values of water tortuosity, and, higher values of "porosity exponent" cause lower values of water tortuosity, because it is in the denominator. A higher value of water tortuosity means that constituents diffusing in water have to travel farther, along a more tortuous path, so the overall rate of diffusion is lower. Figure 10 shows that as the "water content exponent" increases, uranium hazard decreases. Concurrently, as the "porosity exponent" increases, uranium hazard increases. Both of these indicate that as water tortuosity increases, the uranium HQ decreases.

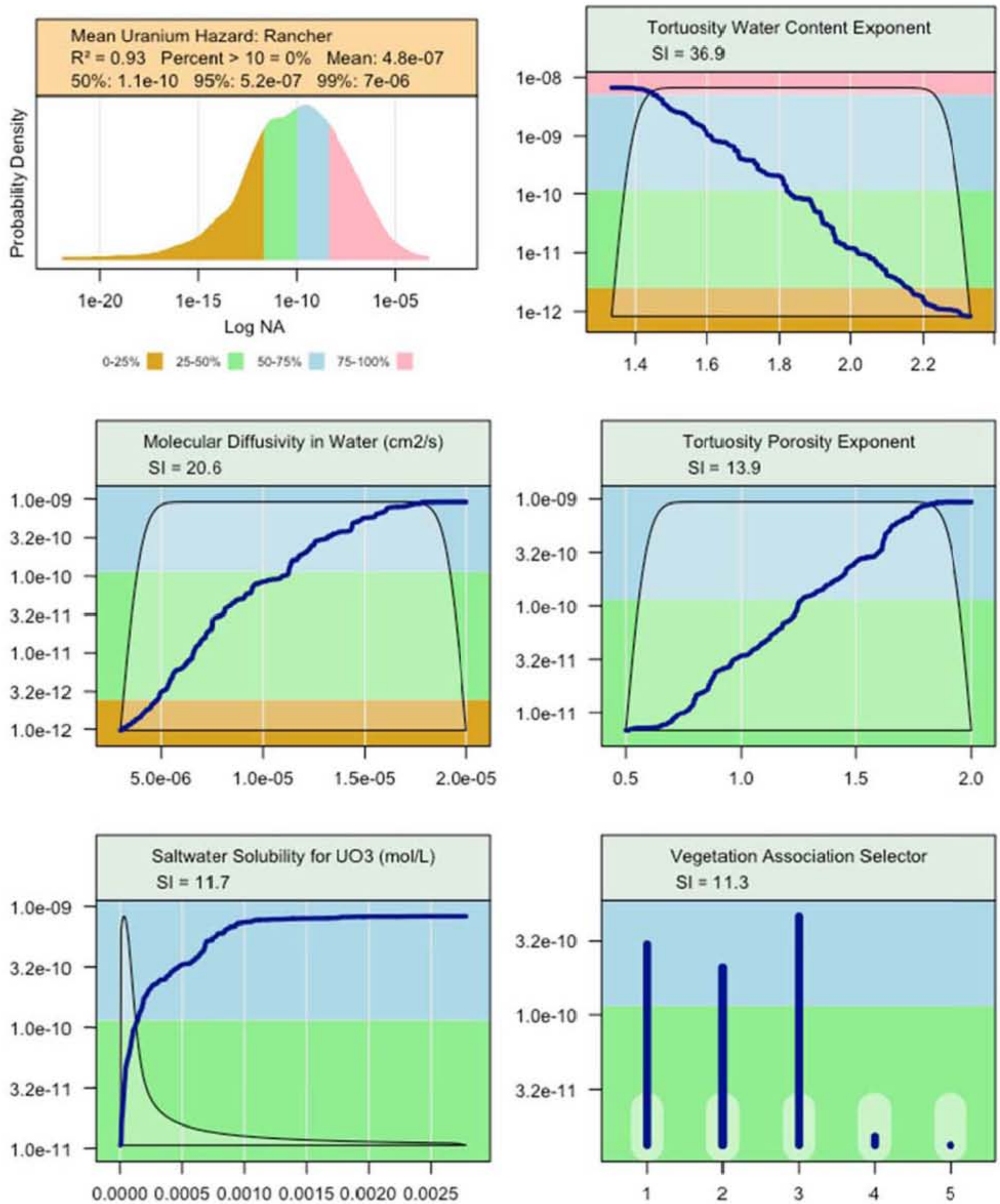


**Figure 10. Partial dependence plots for the mean ranch worker uranium hazard quotient, assuming waste at 3 m and no gullies**

In addition to the effects of tortuosity on diffusion, and ultimately on uranium HQ, the molecular diffusivity $D_m$ in water is also identified as a sensitive parameter. As the diffusivity increases, so does the uranium HQ. Together, these three variables identify water phase diffusion as being positively correlated to the uranium HQ.

Uranium hazard quotient is obviously tied directly to uranium, and no other radionuclides except uranium parents (of which there are very few) could influence this endpoint. The significance of water diffusion indicates that uranium is migrating by diffusion in the water phase. While the bulk of the uranium in the DU waste is $U_3O_8$, and therefore essentially insoluble, the SRS DU is $UO_3$, which is much more soluble. The fourth sensitive variable is the solubility of $UO_3$ in water (all water in the embankment is assumed to be salty water), bolstering the argument that uranium is diffusing in the water phase.

The final piece of this transport and exposure pathway puzzle is hinted at by the fifth-ranked variable: the vegetation associate selector. This stochastic chooses at random from one of five future vegetation associations, each of which is currently found in the vicinity of Clive. These five plots and their vegetation associations are:

- Plot 1: Mixed Grassland,
- Plot 2: Juniper – Sagebrush,
- Plot 3: Black Greasewood,
- Plot 4: Halogeton – Disturbed, and
- Plot 5: Shadscale - Gray Molly.

It is interesting to note that Plot 3 is correlated to the highest values of uranium HQ, implicating black greasewood as a plant of interest. While the black greasewood vegetation association is not entirely made up of that plant type, it is the association with the most black greasewood. Of all the plant types considered in the model, black greasewood has the greatest rooting depth, with a maximum of 5.7 m. With an embankment cover of about 1.65 m, this leaves black greasewood delving about 4 m into the waste below the cover. With the waste emplaced at 3 m below the cover, black greasewood can tap directly into the waste. But even without such deep roots, plants can access uranium that is diffusing upward into the cover. Thus, the following contaminant transport pathway for uranium is suggested: Uranium is leached into infiltrating water, and can diffuse upward to the point that it is within reach of plant roots. With black greasewood in the mix, it does not even have to move upward.

Adding gullies into the mix changes the uranium hazard quotient results significantly for wastes emplaced up to 3 m below the embankment cover. Uranium HQs are much higher, and the variables influencing the value are quite different. As seen in Figure 11, the sensitive parameters are mostly gully-related: Angle of repose of the debris fan is the most significant, describing about two thirds of the variation. This is followed by the gully shape parameter $b$, defining the degree of curvature of the gully thalweg, and thereby related to the depth and volume of the gully. Greater values of $b$ mean larger gullies, leading to increases uranium HQ. The fourth-ranked variable is the angle of repose of the gully walls, also influencing gully outwash volume.

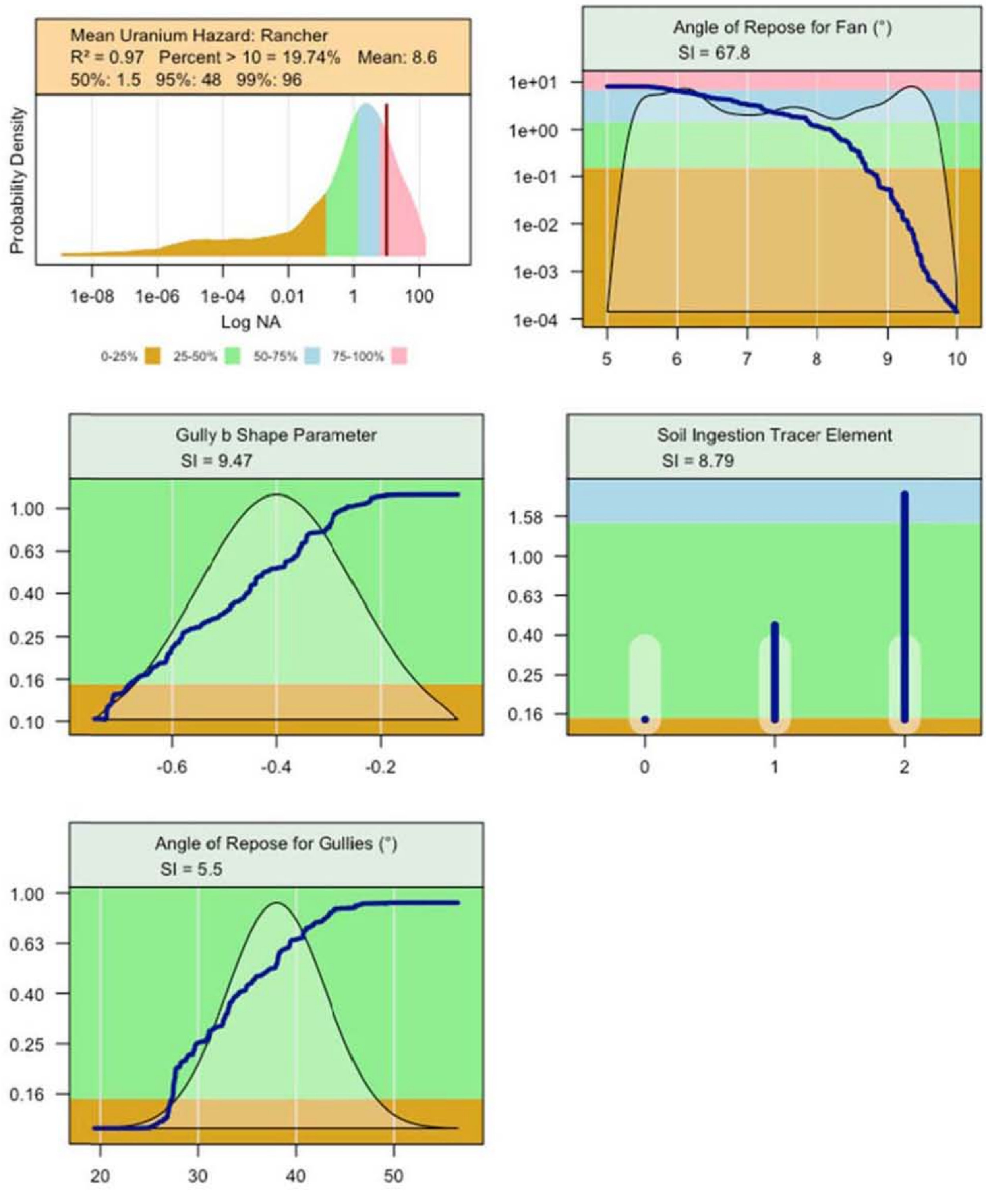


**Figure 11. Partial dependence plots for the mean ranch worker uranium hazard quotient, assuming waste at 3 m, with gullies**

A new variable here is the third one: the selection of the soil ingestion tracer element. In the estimation of soil ingestion rates by humans, as described in the *Dose Assessment* white paper (Appendix 11), various tracers are used, including silicon, aluminum, and titanium. The choice of which of these to use has an effect on this endpoint, but the real message is that soil ingestion is the exposure pathway leading to the uranium exposure.

For the cases where waste is emplaced below 5 m or 10 m, the influences on uranium hazard quotient are much more like those found in the cases where no gullies are considered at all.

A summary of sensitive parameters for each endpoint is provided in the following tables, showing for each of the principal receptors (ranch worker, hunter, and OHV enthusiast) the sensitivity to input parameters for the various waste emplacement depth cases, and with (Table 9) or without (Table 10) gullies. Only those input parameters with a sensitivity index (SI) over 5% are shown.

**Table 9. Sensitivities of peak mean uranium hazard quotient within 10,000 yr, with gully screening calculation**

| receptor | SI rank | input parameter | sensitivity index (SI) |
|---|---|---|---|
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 1 | angle of repose for outwash fan | 68 |
| | 2 | gully thalweg shape parameter b | 9.5 |
| | 3 | soil ingestion tracer element | 8.8 |
| | 4 | angle of repose in gullies | 5.5 |
| hunter | 1 | angle of repose for outwash fan | 68 |
| | 2 | gully thalweg shape parameter b | 9.5 |
| | 3 | soil ingestion tracer element | 9.0 |
| | 4 | angle of repose in gullies | 5.5 |
| OHV enthusiast | 1 | angle of repose for outwash fan | 67 |
| | 2 | gully thalweg shape parameter b | 9.4 |
| | 3 | soil ingestion tracer element | 9.3 |
| | 4 | angle of repose in gullies | 5.4 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 1 | angle of repose for outwash fan | 50 |
| | 2 | water content exponent for tortuosity model | 14 |
| | 3 | aqueous solubility of $UO_3$ | 10 |
| | 4 | molecular diffusivity in water | 9.1 |
| | 5 | porosity exponent for tortuosity model | 6.0 |
| hunter | 1 | angle of repose for outwash fan | 50 |
| | 2 | water content exponent for tortuosity model | 14 |

| | | | |
|---|---|---|---|
| | 3 | aqueous solubility of $UO_3$ | 10 |
| | 4 | molecular diffusivity in water | 9.2 |
| | 5 | porosity exponent for tortuosity model | 6.0 |
| OHV enthusiast | 1 | angle of repose for outwash fan | 49 |
| | 2 | water content exponent for tortuosity model | 14 |
| | 3 | aqueous solubility of $UO_3$ | 10 |
| | 4 | molecular diffusivity in water | 9.2 |
| | 5 | porosity exponent for tortuosity model | 6.0 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 1 | water content exponent for tortuosity model | 38 |
| | 2 | molecular diffusivity in water | 24 |
| | 3 | porosity exponent for tortuosity model | 16 |
| | 4 | angle of repose for outwash fan | 9.4 |
| | 5 | uranium $K_d$ in sand | 7.5 |
| hunter | 1 | water content exponent for tortuosity model | 38 |
| | 2 | molecular diffusivity in water | 25 |
| | 3 | porosity exponent for tortuosity model | 16 |
| | 4 | angle of repose for outwash fan | 9.3 |
| | 5 | uranium $K_d$ in sand | 7.3 |
| OHV enthusiast | 1 | water content exponent for tortuosity model | 38 |
| | 2 | molecular diffusivity in water | 25 |
| | 3 | porosity exponent for tortuosity model | 16 |
| | 4 | angle of repose for outwash fan | 9.3 |
| | 5 | uranium $K_d$ in sand | 7.2 |

**Table 10. Sensitivities of peak mean uranium hazard quotient within 10,000 yr, with no gullies**

| receptor | SI rank | input parameter | sensitivity index (SI) |
|---|---|---|---|
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 1 | water content exponent for tortuosity model | 37 |
| | 2 | molecular diffusivity in water | 21 |
| | 3 | porosity exponent for tortuosity model | 14 |
| | 4 | aqueous solubility of $UO_3$ | 12 |
| | 5 | vegetation association selector | 11 |
| hunter | 1 | water content exponent for tortuosity model | 37 |
| | 2 | molecular diffusivity in water | 21 |

| | | | |
|---|---|---|---|
| | 3 | porosity exponent for tortuosity model | 14 |
| | 4 | vegetation association selector | 12 |
| | 5 | aqueous solubility of $UO_3$ | 12 |
| OHV enthusiast | 1 | water content exponent for tortuosity model | 37 |
| | 2 | molecular diffusivity in water | 21 |
| | 3 | porosity exponent for tortuosity model | 14 |
| | 4 | aqueous solubility of $UO_3$ | 11 |
| | 5 | vegetation association selector | 8.3 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 1 | water content exponent for tortuosity model | 41 |
| | 2 | molecular diffusivity in water | 27 |
| | 3 | porosity exponent for tortuosity model | 16 |
| | 4 | aqueous solubility of $UO_3$ | 5.3 |
| hunter | 1 | water content exponent for tortuosity model | 41 |
| | 2 | molecular diffusivity in water | 27 |
| | 3 | porosity exponent for tortuosity model | 16 |
| | 4 | aqueous solubility of $UO_3$ | 5.3 |
| OHV enthusiast | 1 | water content exponent for tortuosity model | 41 |
| | 2 | molecular diffusivity in water | 26.8 |
| | 3 | porosity exponent for tortuosity model | 17 |
| | 4 | aqueous solubility of $UO_3$ | 5.2 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 1 | water content exponent for tortuosity model | 34 |
| | 2 | molecular diffusivity in water | 23 |
| | 3 | porosity exponent for tortuosity model | 14 |
| | 4 | uranium $K_d$ in sand | 13 |
| hunter | 1 | water content exponent for tortuosity model | 31 |
| | 2 | molecular diffusivity in water | 22 |
| | 3 | porosity exponent for tortuosity model | 13 |
| | 4 | uranium $K_d$ in sand | 12 |
| OHV enthusiast | 1 | water content exponent for tortuosity model | 31 |
| | 2 | molecular diffusivity in water | 22 |
| | 3 | porosity exponent for tortuosity model | 13 |
| | 4 | uranium $K_d$ in sand | 11 |

Like the sensitivity analysis for the dose endpoints, this analysis shows that in the presence of gullies in the 3-m configuration, the parameters defining gully formation are the most important. If wastes are not exposed, as in the cases of deeper waste emplacement, the gullies do not generally encounter wastes, and therefore the outwash is not contaminated, and there is no associated contribution to uranium HQ. Overall, the conclusion of the sensitivity analyses for both the dose and uranium hazard clearly show that the intrusion of gullies into waste-bearing layers has a detrimental effect.

## 6.4 ALARA

In keeping doses as low as reasonably achievable (ALARA) it is necessary to estimate doses not to individuals, but to the entire population of individuals. One such calculation is the cumulative dose to all ranch workers, hunters, and OHV enthusiasts, summed across all individuals and all years of the 10,000-yr simulation. These cumulative population doses, as TEDE, are shown in Table 11, considering the various cases of waste placement and whether the gully screening calculation is included in the analysis.

**Table 11. Peak cumulative population TEDE: statistical summary**

| | Peak population TEDE (rem) within 10,000 yr | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | 35.2 | 29.2 | 87.3 |
| no gullies; waste > 5 m below cover | 4.07 | 3.46 | 9.78 |
| no gullies; waste > 10 m below cover | 0.0434 | 0.0356 | 0.103 |
| with gullies; waste > 3 m below cover | 378 | 172 | 1430 |
| with gullies; waste > 5 m below cover | 4.46 | 3.7 | 10.7 |
| with gullies; waste > 10 m below cover | 0.0448 | 0.0364 | 0.108 |

These population doses represent sum of the average population dose in each year summed over the three classes of receptors. These population doses are very small for five of these six scenarios. A measure for these population doses can be obtained by considering the person-rem costs suggested in NRC and DOE guidance (see the *Decision Analysis* white paper – Appendix 12). Prior to 1995, NRC suggested a flat \$1,000 per person-rem cost. Subsequent to 1995, NRC suggested a value of \$2,000 with a discounting factor of 7% for the first 100 years, and 3% thereafter. NRC also suggested that a range of \$1,000 to \$6,000 might be reasonable, with a best estimate of \$2,000. NRC noted that the intent of raising the person-rem costs from \$1,000 to \$2,000 was to accommodate discounting in an economic analysis. Note that the intent of the NRC approach is to capture the societal effects of added dose to the public.

If a flat rate of \$1,000 is applied to the population dose estimates provided above, then the costs associated with these scenarios are provided in Table 12. These are the total costs over 10 ky. The costs per year are very small, even in the worst case scenario of gullies and a 3-m configuration.

**Table 12. Statistical summary of the flat rate ALARA costs**

| | Peak population ALARA costs within 10,000 yr | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | \$35,000 | \$29,000 | \$87,000 |
| no gullies; waste > 5 m below cover | \$4,000 | \$3,000 | \$10,000 |
| no gullies; waste > 10 m below cover | \$43 | \$36 | \$100 |
| with gullies; waste > 3 m below cover | \$378,000 | \$172,000 | \$1,430,000 |
| with gullies; waste > 5 m below cover | \$4,500 | \$3,700 | \$10,000 |
| with gullies; waste > 10 m below cover | \$45 | \$36 | \$110 |

An approach to discounting could also be applied as suggested by NRC, but this would simply result in lower costs again. For simplicity, assume that the population doses are the same every year, apply the cost of \$2,000 per person rem, and the NRC discount factor. If a discount factor of 7% is applied for the first 100 years, then the ALARA costs are negligible. If a 3% factor is applied across all time, then the total ALARA costs is less than 1% of the undiscounted ALARA costs presented in Table 12 even with the change in starting point from \$1,000 to \$2,000 per person-rem.

In using this approach to ALARA for informed decisions, the ALARA costs involved are very small. The reason the ALARA costs are small is because there are not many receptors in the model that are involved in ranching, hunting or OHV activities at the site. The disposal site is located in a hostile environment, far from most human population centers, and groundwater is not potable. This analysis shows that the number of people engaged in the general vicinity of Clive over the next 10 ky is small, and the doses they are likely to receive are small.

## 6.5 Deep Time Results

The deep time model addresses in a heuristic fashion the fate of the CAS embankment from 10 ky to 2.1 My, the time at which DU reaches secular equilibrium. The model addresses the needs identified in the Section 2(a) of R313-25-8 of the UAC to perform additional simulations for the period where peak dose occurs, for which the results are to be analyzed qualitatively. The deep-time model runs simulations to 2.1 My, but does not calculate dose because of the huge uncertainty in predicting human society and evolution that far into the future, and because the requirement is to analyze simulation results qualitatively. Instead the output of the deep time model is presented in terms of concentrations of radionuclides in relevant environmental media.

The deep-time model considers the return of lakes in the Bonneville Basin that reach or exceed the elevation of Clive. Two classes of lakes are considered. The first is a large lake similar to Lake Bonneville that not only inundates the Clive facility, but also is deep enough and has sufficient duration that lake sedimentation will add to the materials that are currently on Bonneville Basin floor. This type of lake is assumed to occur once every 100 ky in line with the 100-ky climate cycles that have occurred for the past 1 My or so. The second type of lake is shallower, and is termed an intermediate lake. It is also assumed to inundate the Clive facility, but is not a deep lake like Lake Bonneville. It is more similar to the Gilbert Lake that occurred at the end of the last ice age. This type of lake is assumed to occur several times in each climate cycle in response to colder, wetter conditions.

Return of a lake at or above the elevation of Clive is assumed to result in the destruction of the CAS embankment. The above grade embankment material and all of the DU waste is assumed to be dispersed through wave action. The dispersal area forms the basis for the volume of water in which DU waste is dissolved, and ultimately settles back to the basin floor through precipitation or through evaporation as the lake recedes. The lake cycle involves movement of the DU waste, subject to continuing decay and ingrowth, from the sediment into lake water, and back to sediment as the lake forms and recedes. The DU waste is assumed to be fully mixed with the accumulated sediment. Sediment accumulates on average at the rate of about 17 m per 100 ky climate cycle. The current Unit 3 layer of sediment at Clive, which is derived from Lake Bonneville, is assumed to be a confining layer.

The lake cycle effects on transport processes are complex. Sediment core records show significant mixing of sediment, but also can be used to identify significant lake events in the past several hundred thousand years. The extent of sediment mixing is not well understood. The mechanisms for dispersal of a relatively soft pile of material in the middle of a desert flat is not well understood. The extent of mixing of dissolved materials in a large lake is also not well understood. The Model, consequently, is simplified to the point of acknowledging lake return, destruction of the CAS embankment, and cycling of DU waste material between periodic lakes and basin sediments.

In particular, the model overly simplifies the lake cycle processes, and the effect of those processes on the transport of DU waste, and limits the dispersal of DU waste through time. Destruction of the CAS embankment is assumed to occur with a lake that at least reaches the elevation of Clive. This means that even a very shallow lake is assumed to destroy the embankment. It is possible that such lakes, that barely exceed the elevation of Clive would not possess sufficient power to destroy the embankment, and that a different threshold for intermediate lake elevation would be more appropriate. Once the embankment is destroyed, the amount of sedimentation is tied to lake elevation, and the volume of lake water into which DU waste can mix is similarly limited. At the extremes of the input distributions for these factors, some (perhaps unreasonably) high lake water and sediment concentrations are predicted by the Model.

The area of dispersal of the CAS embankment is captured with a simple model that allows the embankment material to spread out according to a specified depth of material that limits the dispersal area. This fixes a dispersal area, but wave action is unlikely to limit the effects of dispersal to such a uniform layer.

Dissolution into the lake is assumed to occur only in the lake volume immediately above the dispersed area. This limits the volume of water within which dissolved materials might mix, and limits the area in which precipitates and evaporates can return. In addition, radon does not escape from the modeled system.

Although the embankment material is dispersed within a dispersal area, isolation of any part of the sediment profile is assumed not to occur. That is, the sediment is assumed to completely mix with previous sediment for every lake event. Lake sedimentation does not allow burial or isolation of previously formed sediment layers. Since different lakes can be identified in sediment cores, this again limits the dispersal of the DU waste.

Initial dispersal of the embankment includes all of the DU waste, even if some of the DU waste is disposed below grade. Return of a lake results in some mixing of sediments with new lake material, but the depth of sediment mixing is not well known. Consequently, all waste is assumed to be mixed with the sediment from the first returning lake. A consequence is that all DU waste in the sediment is available for dissolution into a new lake, no matter how thick the completely mixed sediment.

The model, therefore, represents a closed system that cycles DU waste from lake water to sediment and back again. Decreased concentrations in sediment are obtained because of the increased sediment load, but the mass of DU waste in each lake is not different except from decay and ingrowth.

In light of the simplifications that are included in the model, the results for the deep time scenario are presented for the first 100-ky cycle only, in which the first intermediate or large lake will return and the CAS embankment will be obliterated. The effect of dispersal on concentrations in lake water and sediment are presented for that time frame with a focus on $^{238}$U. Conceptually, deep time will result in a combination of repeated isolation of sediment layers and much greater dispersal than modeled. This will cause mixing over ever increasing areas and volumes, rather than mixing within a closed system. Consequently, concentrations of radionuclides in the DU waste will decrease with each lake cycle and with each climate cycle. However, the constraints of the model do not allow lake water concentrations to decrease with each cycle, and sediment concentrations decrease only because of the additional mass of sediment in which the DU waste is mixed.

The focus of the deep-time results is, consequently, concentrations of $^{238}$U in lake water and sediments within the first 100-ky climate cycle.

### 6.5.1 Lake Water Concentrations of Uranium-238

A summary of lake water concentrations of $^{238}$U is presented in Table 13. Results are presented only for the "no gullies" model scenarios. Based on the model structure, the results should be the same for each of these three models (and also for the cases with gullies if they were included). This is because the entire existing inventory of DU waste is dispersed upon destruction of the embankment, including the waste disposed below grade. The inventory might differ to some relatively small extent because of transport of radionuclides to groundwater or to the accessible environment prior to the return of the lake. However, the lack of systematic differences in the results presented in Table 13 is more suggestive of simulation uncertainty. A time history plot is presented in Figure 12. The jagged nature of the plot is because lake water concentrations are zero when there is no lake present, and intermediate lakes only occur a handful of times prior to formation of the large lake at the end of the 100-ky climate cycle. The peak lake water concentrations occur near the end of the period of the large lake, which provides time for the $^{238}$U to dissolve into the lake.

**Table 13. Statistical summary of peak mean uranium-238 concentrations in lake water within the first 100-ky climate cycle**

| | Peak mean lake water concentration of $^{238}$U within 100 ky (pCi/L) | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | 0.18 | 0.0010 | 1.1 |
| no gullies; waste > 5 m below cover | 0.17 | 0.0009 | 1.0 |
| no gullies; waste > 10 m below cover | 0.18 | 0.0009 | 1.3 |

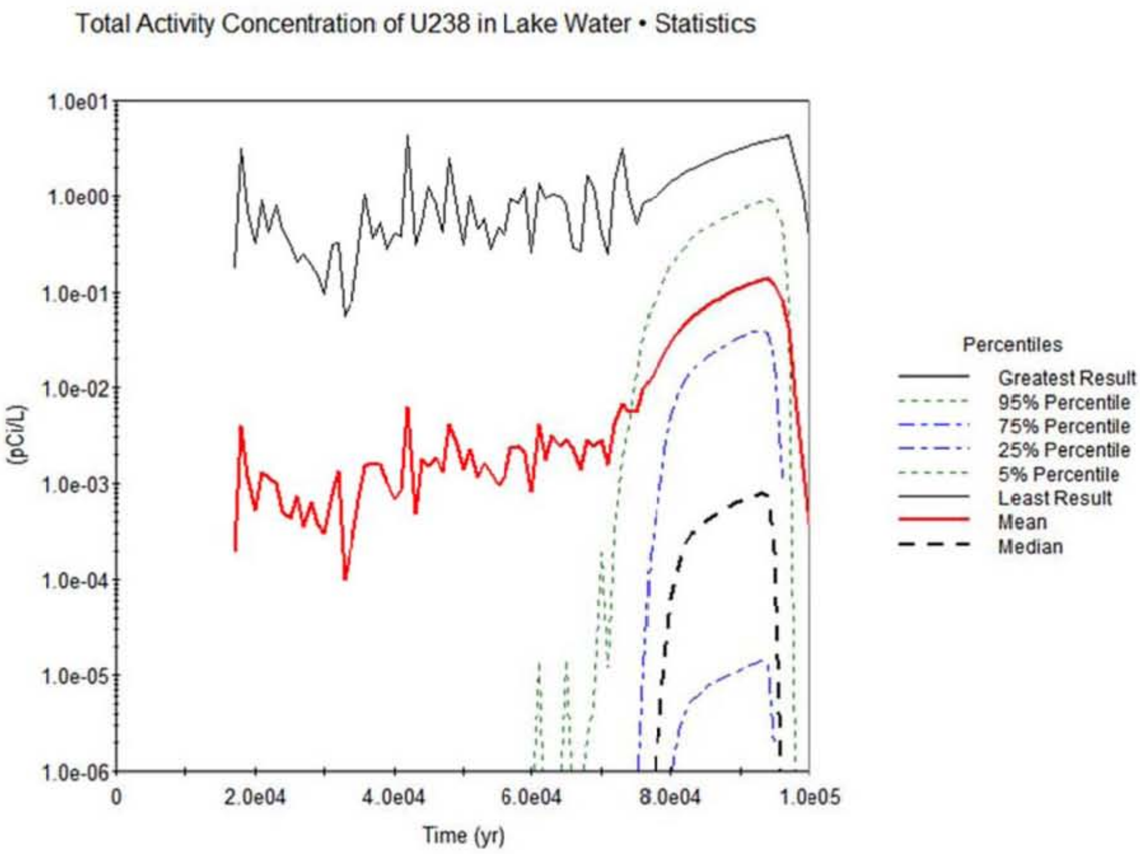


**Figure 12. Time history of mean concentrations of uranium-238 in lake water**

### 6.5.2 Lake Sediment Concentrations of Uranium-238

Results are presented similarly in Table 14 for concentrations of $^{238}U$ in sediment derived from successive lakes. The slight differences are, again due to simulation uncertainty.

**Table 14. Statistical summary of peak mean uranium-238 concentrations in sediment within the first 100-ky climate cycle**

| | Peak mean sediment concentration of $^{238}U$ within 100 ky (pCi/g) | | |
|---|---|---|---|
| **simulation scenario** | **mean** | **median ($50^{th}$ %ile)** | **$95^{th}$ %ile** |
| no gullies; waste > 3 m below cover | 1,600 | 1,300 | 3,600 |
| no gullies; waste > 5 m below cover | 1,500 | 1,300 | 3,400 |
| no gullies; waste > 10 m below cover | 1,500 | 1,300 | 3,400 |

A time history of $^{238}U$ concentrations in future lake sediments is presented in Figure 13. This shows a large increase in concentrations as a consequence of the first lake event, with subsequent decreases as the sediment load increases.

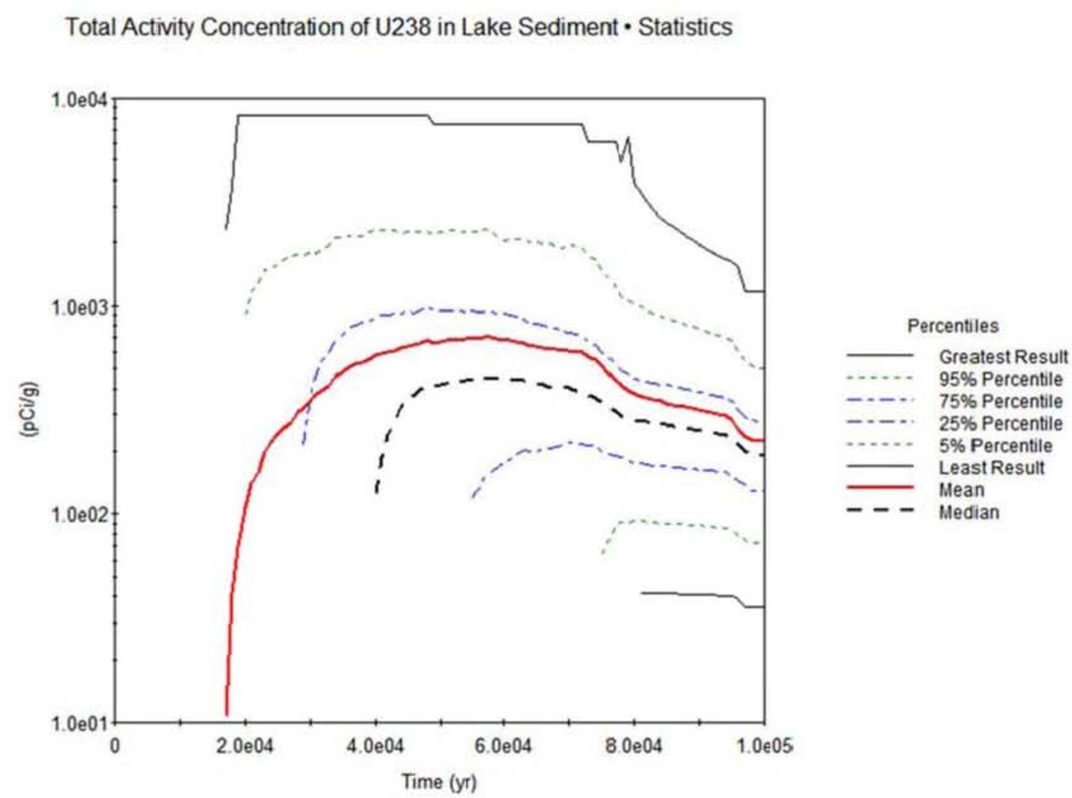


**Figure 13. Time history of mean concentrations of uranium-238 in sediments**

One of the objectives of a PA, as defined in the UAC R313-25-8 is site stability. The performance standard for stability requires the facility must be sited, designed, and closed to achieve long-term stability to eliminate to the extent practicable the need for ongoing active maintenance of the site following closure. If the intent is the need to minimize the need for ongoing active maintenance, as stated, then obliteration of the CAS embankment in deep time achieves this goal. The DU waste material is broadly dispersed with the embankment material, resulting in substantial dilution so that concentrations are low and the need to maintain the site disappears completely.

# 7.0 Summary

This report has laid out the approach taken to developing the PA model for DU waste disposal options at the Clive facility, and has presented results of the initial model (Clive DU PA Model v1.0) with accompanying sensitivity analyses. The purpose of this section is to provide an interpretation of the results in the context of the model, and to compare the results more directly to performance objectives in a compliance evaluation.

## 7.1 Interpretation of Results

Important results of the quantitative PA Model can be summarized, given the compliance time frames of interest, in terms of doses to ranch workers, groundwater concentrations of $^{99}$Tc, and the effect of including gullies in the model. Three waste configurations have been considered, with waste placed at different depths to explore possible disposal configurations. Results of the simplistic ALARA analysis and concentrations in lake water and lake sediments in deep time are also of interest.

Doses to ranch workers increase as the waste is emplaced nearer to the embankment surface in the disposal facility. These doses are driven primarily by exposure to radon. However, groundwater concentrations of $^{99}$Tc increase as the waste is emplaced lower in the disposal facility. These concentrations are driven primarily by the $K_d$ for $^{99}$Tc in sand, but the magnitude of the concentrations is also affected by the concentration distributions used for $^{99}$Tc in the model, and the infiltration rates estimated from the HELP model. That is, the $^{99}$Tc groundwater concentrations could be overestimated. These results highlight the trade-off between disposal configurations that place DU waste higher or lower in the disposal facility. Transport mechanisms move waste either up into the accessible environment or down towards groundwater. A balance is indicated so that performance objectives can be satisfied for these competing endpoints.

The dose results are sensitive to radon. Radon dose assessment is controversial, and takes a different path than dose assessment for other radionuclides, as described in the *Dose Assessment* white paper (Appendix 11).

Once gullies are involved, the doses increase (groundwater concentrations do not change noticeably). However, the effect is most noticeable for the 3-m configuration because sometimes gullies cut into the waste when the waste is placed that high in the embankment. The gully model is a stylized model, developed to examine the potential effects of inclusion of gullies in the Model. The calculation of the depth of gullies is sound, but the number of possible gullies of between 1 and 20 is included in the model only to evaluate sensitivity. The depth of gullies appears as more sensitive than the number of gullies, but both factors are important. Gullies are assumed to be caused by an initiating event such as OHV activity, cattle trails, or biotic impacts (animal burrowing). However, the impact of gullies has not been fully developed in terms of their effect on biotic activity, radon transport, or infiltration.

The ALARA analysis results are interesting mostly because the population doses are very small, which leads to very small ALARA costs, especially if the costs are discounted over time. The population doses are small because the population itself is small, and the doses are also small. Taking this ALARA approach to site performance would suggest that this is a good site for disposal of DU waste. There is room for improvement in this crude ALARA decision analysis. For example, other factors could be included in the analysis such as transportation and worker safety factors, and the cost per person rem could be reevaluated. However, the small population because of the remoteness of the facility, and the low doses suggest that the disposal system would meet ALARA-based performance objectives.

The deep-time model should be regarded as heuristic or highly stylized. Nevertheless, it models the basic concepts of the return of lakes in the Bonneville Basin at or above the elevation of the Clive facility. A sufficiently large lake destroys the DU disposal facility, redistributes DU waste with the lake sediment, and repeats the cycles of DU waste moving into lake water, and settling back into sediment. Sedimentation rates are about 17 m per 100 ky, and the DU waste is assumed to mix with the sediment across time. There are several components of this heuristic model that could be regarded as conservative in the sense of over predicting concentration in both lake water and lake sediment. For example, all of the DU waste that is still in the disposal system is assumed to be dispersed when the embankment is obliterated, even though it might be reasonable to assume that the waste disposed below grade would be covered by lake sediment. Also, in the model a lake can destroy the site when it reaches the Clive elevation, which can cause mixing of waste in a very shallow lake, a lake that perhaps does not have sufficient power to destroy the facility. Research into the power needed for a lake to destroy the facility might indicate the minimum elevation needed for such an event. The embankment is dispersed over a comparatively small area in some simulations. Research into physical dispersal as a consequence of lake-induced destruction might be revealing. A water column is assumed above the dispersal area, which limits the amount of water available for mixing with DU waste. And, sediment mixing is assumed to occur with every lake cycle, even though some lake cycles might bury some sediment. Despite these possible conservatisms in the deep-time model, the lake water and lake sediment concentrations are small. They reflect concentrations associated with the first lake event, consistent with the timing of the maximum lake water and lake sediment concentrations.

Lake water concentrations of $^{238}U$ in the first 100-ky climate cycle average less than 1 pCi/L, even given the conservatism in the model. The peak of the mean concentrations of $^{238}U$ in sediment average about 1,500 pCi/g, with a 95$^{th}$ percentile of about 3,500 pCi/g. Given the simplified model structure, these lake water and sediment concentrations are probably considerable overestimates, and the concentrations should decrease with time as a consequence of further dispersal of the DU waste with other material over time.

## 7.2 Comparison to Performance Objectives

Comparisons to performance objectives are presented for doses to ranch workers, since dose to other receptors are considerably less, and groundwater concentration for $^{99}Tc$. The evaluations address the three disposal configuration scenarios (3-m, 5-m, and 10-m) and exclusion/inclusion of gullies. Quantitative performance objectives do not exist for the ALARA analysis or for the deep-time concentrations endpoints.

The concentrations reported by the PA model represent estimates of the mean concentration in each year. The peak of those mean concentrations is collected across the 500-yr compliance period. Because the groundwater concentration of $^{99}$Tc increases with time, the peak of the mean concentration occurs at 500 yrs. The 5,000 simulations provide 5,000 estimates of the peak of the mean concentrations. Summary statistics for the distribution of the peak of the mean $^{99}$Tc concentrations are presented in Table 15. For the 3-m and 5-m models, compliance with the GWPLs is clearly demonstrated. For the 10-m model the situation is not as clear. However, both the mean (of the peak of the means) and the 95$^{th}$ percentile exceed the GWPL, in which case, it is probably reasonable to conclude that the 10-m scenario is not in compliance with the performance objective.

The results depend critically on the model structure, specification and underlying assumptions. Infiltration rates might be overestimated, and $^{99}$Tc inventory concentrations might be overestimated. However, based on the model assumptions the 10-m model does not comply with the GWPL performance objective for $^{99}$Tc. These results suggest that there are configurations that comply with the GWPLs.

**Table 15. Peak groundwater activity concentrations for $^{99}$Tc within 500 yr, compared to GWPLs**

| radionuclide | GWPL (pCi/L) | peak activity concentration within 500 yr (pCi/L) | | |
|---|---|---|---|---|
| | | mean | median (50$^{th}$ %ile) | 95$^{th}$ %ile |
| waste emplaced > 3 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 85.9 | 1.43e-5 | 209 |
| waste emplaced > 5 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 437 | 0.00264 | 1710 |
| waste emplaced > 10 m below embankment cover | | | | |
| $^{99}$Tc | 3790 | 14400 | 113 | 81400 |

The dose results for ranch workers are presented in Table 16 for the no gully scenario, Table 17 for the scenario with gullies, and for all three disposal configurations. The statistics represent summaries of the peak of the mean doses. Considering that doses increase with time given the model construction and assumptions, then the 95$^{th}$ percentile is analogous to the 95% upper confidence interval of the mean that is common in CERLCA risk assessments. The mean and the 95$^{th}$ percentile are compared to the performance objectives.

The doses increase as waste is placed nearer the top of the embankment, but the MOP performance objectives are not exceeded in all cases. This implies that disposal configurations exist, under the conditions of this model, for which it is reasonable to dispose of DU waste.

**Table 16. Peak mean TEDE, without consideration of gullies: statistical summary**

| | Peak TEDE (mrem in a yr) within 10,000 yr | | |
|---|---|---|---|
| **receptor** | **mean** | **median (50th %ile)** | **95th %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 4.37 | 3.44 | 11.3 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.598 | 0.473 | 1.52 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00596 | 0.00471 | 0.0152 |

**Table 17. Peak mean TEDE, with gully screening calculation: statistical summary**

| | Peak TEDE (mrem in a yr) within 10,000 yr | | |
|---|---|---|---|
| **receptor** | **mean** | **median (50th %ile)** | **95th %ile** |
| waste emplaced > 3 m below embankment cover | | | |
| ranch worker | 20.9 | 11.6 | 72.3 |
| waste emplaced > 5 m below embankment cover | | | |
| ranch worker | 0.564 | 0.443 | 1.44 |
| waste emplaced > 10 m below embankment cover | | | |
| ranch worker | 0.00594 | 0.00457 | 0.0155 |

# 8.0 Conclusions

Model results are dependent on the model structure, specification and assumptions upon which it is based. With the expertise and assumptions upon which the Clive DU PA Model v1.0 is based, the Model demonstrates that there are disposal configuration options for the subject DU waste that are adequately protective of human health and the environment as projected for the next 10,000 years. Protectiveness is assessed under Utah Administrative Code R313-25-8 Section 2(a) by consideration in this PA Model of:

- dose to site-specific receptors,
- concentrations in groundwater,
- ALARA, and
- considerations of deep-time scenarios.

The model was run using three different configurations: 3 m, 5 m, and 10 m of extra fill material. It was also run with and without expectation of gullies forming. Simplified results for these scenarios are presented in Table 18.

**Table 18. Summary of results of the Clive DU PA Model**

| | without gullies: top of waste at | | | with gullies: top of waste at | | |
|---|---|---|---|---|---|---|
| **performance objective** | **3-m** | **5-m** | **10-m** | **3-m** | **5-m** | **10-m** |
| Dose to MOP below regulatory threshold of 25 mrem/year | Yes | Yes | Yes | Maybe[1] | Yes | Yes |
| Dose to IHI below regulatory threshold of 500 mrem/year | Yes | Yes | Yes | Yes | Yes | Yes |
| Groundwater maximum concentration of $^{99}$Tc in 500 years < 3790 pCi/L[3] | Yes | Yes | No[2] | Yes | Yes | No[2] |
| ALARA average total population cost equivalent over 10,000 years | $35,000 | $4,000 | $43 | $378,000 | $4,500 | $45 |

[1]The expected dose to MOP is acceptable under this scenario, but the 95$^{th}$ percentile of the expected dose exceeds the regulatory threshold.

[2]These results might overestimate groundwater concentrations because of potential overestimation of infiltration rates and of the $^{99}$Tc inventory.

[3]Groundwater concentrations of all other radionuclides are significantly less than their respective GWPLs, with the exception of $^{129}$I inventory which is evaluated separately.

The three configurations that are evaluated for the Clive DU PA Model v1.0, with and without consideration of gullies, demonstrate that the disposal facility can adequately protect human health and the environment when disposing of the subject DU waste:

- all disposal options evaluated exhibit doses that are less than the inadvertent intrusion performance objective,
- there are clearly disposal configurations for which the predicted doses are less than the MOP performance objective, and
- there are clearly disposal options for which groundwater concentrations do not exceed GWPLs.

In addition, the ALARA analysis indicates that ALARA costs from population doses that might be realized for the duration of the 10 ky model are very small. On a per year basis, the ALARA costs are always less than $1 per day.

The deep-time model indicates that concentrations in media such as lake water and sediment will continue to decrease with each lake and climate cycle, and that destruction of the site will lead to dispersal of the DU waste in the Bonneville Basin. The CAS embankment will be destroyed and buried by the return of a large lake, but long-term maintenance will be unnecessary.

All conclusions depend on the model structure, specification and assumptions. Changes in any aspect of the model could cause different results.

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# List of Appendices