STATE OF TENNESSEE
DEPARTMENT OF ENVIRONMENT AND CONSERVATION
Division of Remediation - Oak Ridge
761 Emory Valley Road
Oak Ridge, Tennessee 37830

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Dr. Justin Marble
EM-4.21 Co-Chair, Low-Level Waste Disposal Facility Federal Review Group
U.S. Department of Energy
19901 Germantown Road
Germantown, Maryland 20874-1290

Mrs. Sherri Ross
EM-4.31 Co-Chair, Low-Level Waste Disposal Facility Federal Review Group
U.S. Department of Energy
19901 Germantown Road
Germantown, Maryland 20874-1290

Review of Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee

Dr. Marble and Mrs. Ross

As discussed on September 17 and 24, the Tennessee Department of Environment & Conservation (TDEC) offers the enclosed review of the Performance Assessment (PA) and Composite Analysis (CA) for the proposed Environmental Management Disposal Facility (EMDF) in Oak Ridge (Enclosure 1). TDEC contracted Neptune and Company, Inc. (Neptune) as subject matter experts to develop this review, as agreed with the U.S. Department of Energy's (DOE's) Oak Ridge Office of Environmental Management (OREM) in 2017.

Neptune prepared the review to inform TDEC's understanding of the PA and CA as they relate to forthcoming TDEC decisions under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). TDEC agreed to share the review with DOE's Low-Level Waste Disposal Facility Federal Review Group (LFRG) to support discussion of TDEC concerns regarding CERCLA waste acceptance criteria (WAC) to be derived from those evaluations. TDEC appreciates the role LFRG played in reviewing the PA and CA. We also appreciate your willingness to meet with us last month to begin a dialog regarding TDEC's concerns.

WAC will determine what can go into the mixed-waste landfill and what must be sent for treatment and disposal at approved facilities in the western U.S. It is incumbent on DOE to document how waste disposal at EMDF will protect public health, as required by the CERCLA.

From 2012 through 2017, TDEC commented on WAC presented in several draft remedial investigation/feasibility study (RI/FS) reports. Analytic WAC ranged from a U-238 value so low that it would limit the usefulness of an onsite landfill to unlimited values for radionuclides expected to be in the waste, such as cesium-137 and iodine-129. Although TDEC comments were never resolved in a final report, OREM agreed in early 2018 to provide protective WAC before final approval of a Record of Decision (ROD) that selects the remedy of onsite disposal under the Federal Facilities Agreement (FFA). OREM reiterated this commitment in the language of the Proposed Plan published in the fall of 2018. Toward that end, TDEC appreciates OREM's cooperation in supporting independent review of computer modeling in the PA/CA evaluations.

TDEC is interested in understanding how DOE will evaluate, monitor, and mitigate potential risks to human health posed by the toxicity of uranium and hazardous contaminants like mercury and polychlorinated biphenyls (PCBs). This includes evaluation of reasonable scenarios, such as consumption of fish that bioaccumulate contaminants like mercury and some radionuclides. Although TDEC has no regulatory authority over the PA or CA, which are completed under DOE's self-regulatory authority under the Atomic Energy Act of 1954 (AEA), TDEC needs a better understanding of several aspects of the PA/CA modeling if DOE plans to use a similar approach to evaluate CERCLA risk scenarios.

TDEC will require the proposed landfill to have WAC demonstrated to be protective as required by CERCLA. As a starting point for future discussion, Enclosure 2 presents TDEC's comments on draft WAC provided before completion of the PA/CA. In the end, TDEC believes DOE must provide information documenting WAC protectiveness to the public before signature of a ROD because this information was not available in the 2018 Proposed Plan.

Please direct any questions or comments regarding this letter to Brad Stephenson at (865) 220-6587.

Sincerely

Randy C. Young
TDEC FFA Manager

Enclosures

ec: Mike Higgins, DoR
Colby Morgan, DoR
Steve Sanders, DoR
Steve Stout, OGC
Pat Flood, TDEC
Chris Thompson, TDEC
Connie Jones, EPA
Carl Froede, EPA
Jon Richards, EPA

ec: Bill O'Steen, EPA
Roger Petrie, DOE
Pat Halsey, DOE
Dennis Mayton, DOE
OREM Mailroom
Susan DePaoli, Pro-2-Serve
John Patterson, UCOR
Sid Garland, UCOR
Tanya Salamacha, UCOR

xc: Amanda Daugherty, ORRCA
Amy Fitzgerald, ORRCA
Ron Woody, ORRCA
Shelley Kimel, ORSSAB

Enclosure 1

NAC-0131_R1

A Review of the Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee

12 October 2020

Prepared for

Tennessee Department of Environment and Conservation

Prepared by

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

1. Title: A Review of the Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee
2. Filename: Neptune EMDF PACA R2 Review 2020-10-12.docx
3. Description: This is a review of the Performance Assessment and Composite Analysis (PA/CA) for the proposed Environmental Management Disposal Facility (EMDF) on Oak Ridge Reservation, specifically addressing the revised (R2) 2020 PA/CA.
4. Remarks Revision 1 to address revisions to the EMDF PA/CA (R2). Revision 0 focused on the EMDF PA/CA (R1).

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ACRONYMS AND ABBREVIATIONS

ALARA	As Low As Reasonably Achievable
ASME	American Society of Mechanical Engineers
BCBG	Bear Creek Burial Grounds
BCV	Bear Creek Valley
CA	Composite Analysis
CBCV	Central Bear Creek Valley
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act (Superfund)
DCH	(RESRAD) Data Collection Handbook
DCS	Derived Concentration Standard
DOE	(United States) Department of Energy
ELCR	excess lifetime cancer risk
EM	DOE’s Office of Environmental Management
EMDF	Environmental Management Disposal Facility
EMWMF	Environmental Management Waste Management Facility
EOW	edge of waste
E/P ratio	escape-to-production ratio (for radon emanation)
EPA	(United States) Environmental Protection Agency
FEPs	features, events, and processes
FEPSs	features, events, processes, and scenarios
FFA	Federal Facilities Agreement
HELP	Hydrologic Evaluation of Landfill Performance (computer model)
IC	institutional control
IHI	Inadvertent Human Intrusion
LDR	Land Disposal Restrictions
LFRG	Low-Level Waste Disposal Facility Federal Review Group
LLW	low-level (radioactive) waste
MCL	Maximum Concentration Limits
MEI	maximally exposed individual
MOP	member of the public
NQA	nuclear quality assurance
OAT	one-at-a-time
OREM	Oak Ridge Office of Environmental Management
ORNL	Oak Ridge National Laboratory
ORR	Oak Ridge Reservation
PA	Performance Assessment
PCB	polychlorinated biphenyl
POA	point of assessment
RCRA	Resource Conservation and Recovery Act
RESRAD	RESidual RADiation (computer program)
RI	Remedial Investigation
RI/FS	Remedial Investigation/Feasibility Study
ROD	Record of Decision
SA	sensitivity analysis

SSAB	(Oak Ridge) Site Specific Advisory Board
TDEC	Tennessee Department of Environment and Conservation
TSCA	Toxic Substances Control Act
UA	uncertainty analysis
UCOR	URS/CH2M Oak Ridge LLC
WAC	waste acceptance criteria

Executive Summary

On behalf of the Tennessee Department of Environment and Conservation (TDEC), Neptune and Company, Inc. (Neptune) conducted a review and evaluation of the *Performance Assessment for the Environmental Management Disposal Facility at the Y-12 National Security Complex, Oak Ridge, Tennessee* (PA) and the *Composite Analysis for the Environmental Management Waste Management Facility and the Environmental Management Disposal Facility, Oak Ridge, Tennessee* (CA). The purpose of the PA and CA is to demonstrate, using risk assessment methods, that proposed radioactive waste disposal operations will be in compliance with DOE Order 435.1, Radioactive Waste Management.

The PA models hypothetical scenarios that could result in exposure of members of the public to radiation and evaluates the potential for the proposed Environmental Management Disposal Facility (EMDF) to impact human health and water resources. The CA considers the cumulative effects of all potential sources of radioactive contamination within Bear Creek Valley, the proposed site for the EMDF, on health and the environment. Potential sources include the proposed facility; a currently active disposal site for radioactive waste, the Environmental Management Waste Management Facility (EMWMF); and contamination due to legacy disposal operations in the Bear Creek watershed.

This review addresses the April 2020 revisions of the PA and CA, and the modeling approaches as presented. It is prepared for TDEC with specific goals of:

- Explaining the PA/CA process
- Reviewing the EMDF PA/CA
- Highlighting major technical concerns

It is not written with the intent of providing formal comments intended for DOE. It is written more as a description of the process and issues found, rather than specifically as a document that contains specific review comments. Nevertheless, this review identifies and describes the major technical concerns with the EMDF PA/CA that merit consideration before important decisions are made regarding the proposed EMDF.

Overall, the modeling approach used in the EMDF PA would be more effective and defensible if run as a fully probabilistic model. This approach is commonly taken in other DOE PAs using software platforms such as GoldSim, which also fully integrates and couples all relevant transport processes with the dose (risk) assessment. A fully probabilistic model should also include a more appropriate groundwater model for this site (one that can handle flow through fractured media and is calibrated against site data) and more thorough handling of cover degradation processes and bathtubbing coupled with the hydrogeologic modeling that is the current focus of the PA. Decoupling of interrelated phenomena, such as cover degradation, infiltration rate, and bathtubbing, can cause potentially important interactions to be overlooked. The EMDF PA is an outlier among recent PA work (e.g. DOE sites at NNSS, LANL, SRS, Hanford, and West Valley, and private radioactive wastes sites in Texas, Utah, and Idaho) in its decoupled, deterministic modeling and one-at-a-time approach to sensitivity analysis.

This report presents critical issues and key findings identified for discussion with TDEC during Neptune's review and evaluation of the PA and CA. Critical issues are defined as those issues that undermine the conclusions of the PA or CA, while key findings provide further technical details regarding the basis for the identification of critical issues. A number of critical issues are common to both the PA and CA, and fit broadly into several categories:

- **The EMDF PA "base case" radionuclide transport and dose assessment modeling is bounded by assumptions rather than structured to evaluate mechanistic modeling of all applicable events and processes.** This leads to inaccurate and incomplete modeling based on these constraining assumptions. Natural processes that will compromise the ability of the EMDF to isolate contaminants from the environment are either not incorporated into the base case modeling (e.g. gully erosion, "bathtubbing") or they are artificially constrained without supporting rationale (e.g. a twofold linear increase in infiltration up to year 1000, and no further cover degradation after that time). For example, a plausible mechanism leading to release of contaminants is a localized breach of containment at the top of the liner due to accumulation of water in the facility. A release resulting from this mode of failure, often referred to as bathtubbing, seems probable sometime during the compliance period specified by DOE, and such a scenario is considered in some detail in the PA's supporting documentation. Although modeling of this 'bathtub scenario' predicts unacceptable levels of radionuclides in groundwater at a point of assessment 100 meters from the edge of the landfill, this analysis is kept outside of the PA and the results are not used to evaluate facility performance.

- **Contaminant fate and transport modeling does not adequately represent the natural system.** The PA does not address plausible fate and transport pathways including groundwater fracture flow, sheet and gully erosion of the cover, uptake of subsurface radionuclides by deep-rooted plants, and deposition of radon progeny in the cover from the upward diffusion of radon. One example is underprediction of times of travel for contaminants in groundwater. Studies conducted over decades in Oak Ridge have shown that many radionuclides migrate readily through the fractured rocks in Bear Creek Valley. The errors made in solute transport modeling result in the PA's conclusion that a member of the public consuming water or fish in the vicinity of the facility throughout the next millennium would receive a radiation dose from just one isotope, Carbon-14. The transport models should be calibrated using available results from the many field scale tracer tests that have been conducted in Oak Ridge and supplemented with models that incorporate the physics of solute transport in fractured media. Model predictions should be checked against Oak Ridge environmental monitoring data that yield independent estimates of travel times for many radionuclides.

- **The hydrogeologic contaminant transport processes that are modeled are not coupled with other contaminant transport processes.** This problem stems from using software that is not capable of coupling such systems. For example, the upward migration of radon and its progeny (and indeed its parents) is not coupled with the downward transport to groundwater. In nature, these processes occur simultaneously, so decoupling them can cause obscure potentially important interactions.

- **The lack of a fully probabilistic analysis misrepresents what may be important drivers in the analysis.** The "base case" for this assessment is a single deterministic calculation, affording no insight about the context of uncertainty. While a handful of select parameters are used in one-at-a-time sensitivity analysis calculations, these are selected based on their expected significance. Only a fully probabilistic analysis, where all model inputs reflect the uncertainty in their values, would reveal those parameters that have unexpected significance.

The PA and CA do not adequately address other significant issues. Non-radioactive hazardous and toxic wastes to be disposed in the EMDF are not considered in the analysis, even though toxicity from mercury and uranium has the potential to pose a greater risk to human health and water resources than the effects of radiation. Current sources of contamination other than the EMDF and the EMWMF present in Bear Creek Valley are assumed to be remediated for the purposes of the CA, but the practicability of reaching remediation goals for these sources, which contain an inventory of about 20 million kilograms of uranium, is not evaluated. Uncertainties in the inventory of radionuclides in the waste to be disposed at the EMDF are acknowledged, but are not addressed in probabilistic modeling.

Consequently, the conclusions in the PA and CA are not adequately supported. The probability that the proposed facility will contribute to unacceptable levels of contamination in groundwater and surface water in the Bear Creek watershed is potentially greater than the PA and CA suggest. Limits on waste acceptance for the proposed facility, if and when it is authorized, should be based on an analysis that addresses a wider range of underlying assumptions and fully probabilistic modeling.

1.0 Introduction

On behalf of the Tennessee Department of Environment and Conservation (TDEC), Neptune and Company, Inc. (Neptune) reviewed the *Performance Assessment for the Environmental Management Disposal Facility at the Y-12 National Security Complex, Oak Ridge, Tennessee* (UCOR-5094/R1), dated June 20, 2018 (UCOR 2018a), and UCOR-5094/R2, dated April 23, 2020 (UCOR 2020b), and the *Composite Analysis for the Environmental Management Waste Management Facility and the Environmental Management Disposal Facility, Oak Ridge, Tennessee* (UCOR-5095/R1), dated July 11, 2018 (UCOR 2018b), and UCOR-5095/R2, dated April 16, 2020 (UCOR 2020a). This review supports the State of Tennessee's oversight role in accordance with the *Federal Facility Agreement for the Oak Ridge Reservation* (DOE 1996). The review was conducted in consideration of DOE Order (O) 435.1 *Radioactive Waste Management* (DOE 2001a), and DOE Manual (M) 435.1-1 (DOE 2001b).

Neptune also attended regular calls and on-site meetings with TDEC, the U.S. Environmental Protection Agency, the U.S. Department of Energy's (DOE) Oak Ridge Office of Environmental Management (OREM), and OREM contractors and subcontractors, including URS/CH2M Oak Ridge LLC (UCOR). During these calls and meetings, key elements of the Performance Assessment (PA) and Composite Analysis (CA) for the proposed Environmental Management Disposal Facility (EMDF) were presented to DOE's Office of Environmental Management (EM) Low-Level Waste Disposal Facility Federal Review Group (LFRG). The LFRG supports DOE implementation of its regulatory responsibility under the Atomic Energy Act of 1954, as amended, and O 435.1.

These orders and analyses are specific to radiological constituents, over which DOE has self-regulation authority. The EMDF is also proposed to contain significant amounts of wastes containing constituents for which non-radiological toxicity criteria are published, such as uranium (U), mercury (Hg), and polychlorinated biphenyls (PCBs) that are under the purview of TDEC and the U.S. Environmental Protection Agency (EPA). This review is restricted to the PA and CA, neither of which includes an analysis of non-radiological health risks.

This report presents critical issues and key findings identified during Neptune's review and evaluation of the PA and CA. While reviewing R1 of the PA, Neptune conducted supplemental modeling to assess certain aspects of the critical issues. These include evaluations of the "bathtub scenario[1]" presented in Appendix C, Section C.3 of the PA, the Resident Inadvertent Human Intrusion (IHI) scenario presented in Section 6 and Appendix I of the PA, and the potential impacts (fate and concentrations) of non-radiological contaminants (specifically, mercury). Neptune also performed a supplemental evaluation of the radon fate and transport modeling and results presented in the EMDF PA (Appendix H of the PA). Each of these four supplemental evaluations is provided as an appendix (Appendices A, B, C, and D, respectively). Note that Appendices A, B, C, and D have not been specifically updated for the R2 PA; though many of their conclusions remain valid.

[1] A situation where cover components allow precipitation to infiltrate more rapidly than liner components allow it to drain, leading to accumulation of infiltrating water and saturation of the waste.

Neptune identified a number of issues and assumptions used to develop the PA and CA that raise concerns about whether the PA and CA meet the guidelines established in DOE Orders 435.1 and 458.1; and, therefore, whether these evaluations support a determination that the proposed EMDF provides a remedial alternative that is protective of human health and the environment.

2.0 Summary of Findings

This review identifies critical issues and key findings. A critical issue is defined as an issue that affects the veracity of the PA or CA, especially the primary conclusion that in both cases the performance metrics will be met. Key findings are those findings that, either together or individually, provide the basis for the identification of critical issues. Additional key findings regarding the assumptions and methods used to develop the PA and CA not directly related to critical issues are also summarized.

2.1 Critical Issues—Performance Assessment

The EMDF PA "base case" radionuclide transport and dose assessment modeling is bounded by assumptions rather than structured to evaluate mechanistic modeling of all applicable events and processes. Specifically, the performance of the EMDF for the 1000-year and 10,000-year performance periods is controlled by poorly supported and optimistic assumptions.

The PA produced the following results: the peak all-pathways total base case dose is projected to be 1.0 mrem/yr within the 1000-year compliance period, with a peak of 9.1 mrem/yr at 5100 years for the period between 1000 and 10,000 years. All radionuclide dose is related to downward leaching of radionuclides due to infiltration of meteoric water through the EMDF landfill, followed by lateral transport in groundwater, and exposure via pathways related to a domestic groundwater supply well and Bear Creek surface water.

The RESidual RADiation (RESRAD)-OFFSITE model used to generate these base case results is highly dependent on several critical assumptions:

i. The physical integrity of the EMDF landfill cover will not be significantly compromised during the 1000-yr and 10,000-yr time periods evaluated. Specifically, with respect to the groundwater exposure pathways, the long-term degradation of the engineered cover will result in no more than approximately a twofold increase in the original infiltration rate (excluding the geomembrane barrier) during the model evaluation time periods.

ii. Radionuclide mobility is realistically estimated for all domains of the transport model, including waste, vadose, and saturated zones, and will remain consistent over time.

iii. The rate of infiltration through the cover will never exceed the rate of drainage through the liner system, so bathtubbing will not occur. No leakage from the facility will ever occur except that passing through the liner system.

iv. Chronic exposure of potential future receptors will never occur on the EMDF facility. The base case all-pathways model does not consider the possibility of human intrusion into the facility or human occupants on the facility. Inadvertent human intrusion (IHI) is considered in a separate analysis, but the maximum exposure in IHI scenarios is limited to dose resulting from a garden contaminated with drill cuttings from a well drilled into waste.

The PA's Executive Summary provides three key assumptions for PA compliance, and a second set of five key conceptual model assumptions. Collectively, the four critical assumptions highlighted above by Neptune encompass the PA's eight assumptions.

Specific critical issues related to the model assumptions discussed above are detailed in the following sections.

2.1.1 PA Critical Issue 1: Conceptual Model Assumptions

The results of the base case and probabilistic transport and dose modeling performed for the PA are primarily controlled by assumptions that constrain the behavior of the engineered system over time, rather than by modeling how the system might realistically be compromised over time. Without active institutional controls in perpetuity at the site, these assumptions are unlikely to remain valid. In addition, modeling of radionuclide transport outside the engineered system is inconsistent with travel times inferred from environmental monitoring and field studies conducted elsewhere on the Oak Ridge Reservation in the geologic formations underlying the proposed site.

Figure 5.9 of the R1 PA (see Figure 1) indicates that the uncertainty in future radiation dose (measured as the difference between medians or means and 95th percentiles) over a modeling period of 10,000 years is believed to be approximately a factor of three. This degree of confidence in the very long-term performance of the EMDF reflects a concern with the PA modeling—the behavior of the system is controlled by assumptions (model boundaries) rather than by mechanistic modeling of all applicable events and processes. As shown in Figure 2 below, in the R2 PA uncertainty in landfill performance increases significantly in the distant future, although degradation of performance is still heavily constrained by the critical assumptions discussed above. Absent such constraints, the timing of potential doses above 25 mrem/yr would be expected to shift to earlier dates.

Over the 10,000-yr modeling period, the difference in all-pathways probabilistic dose results from the R1 PA to the R2 PA is illustrated in the following figures:

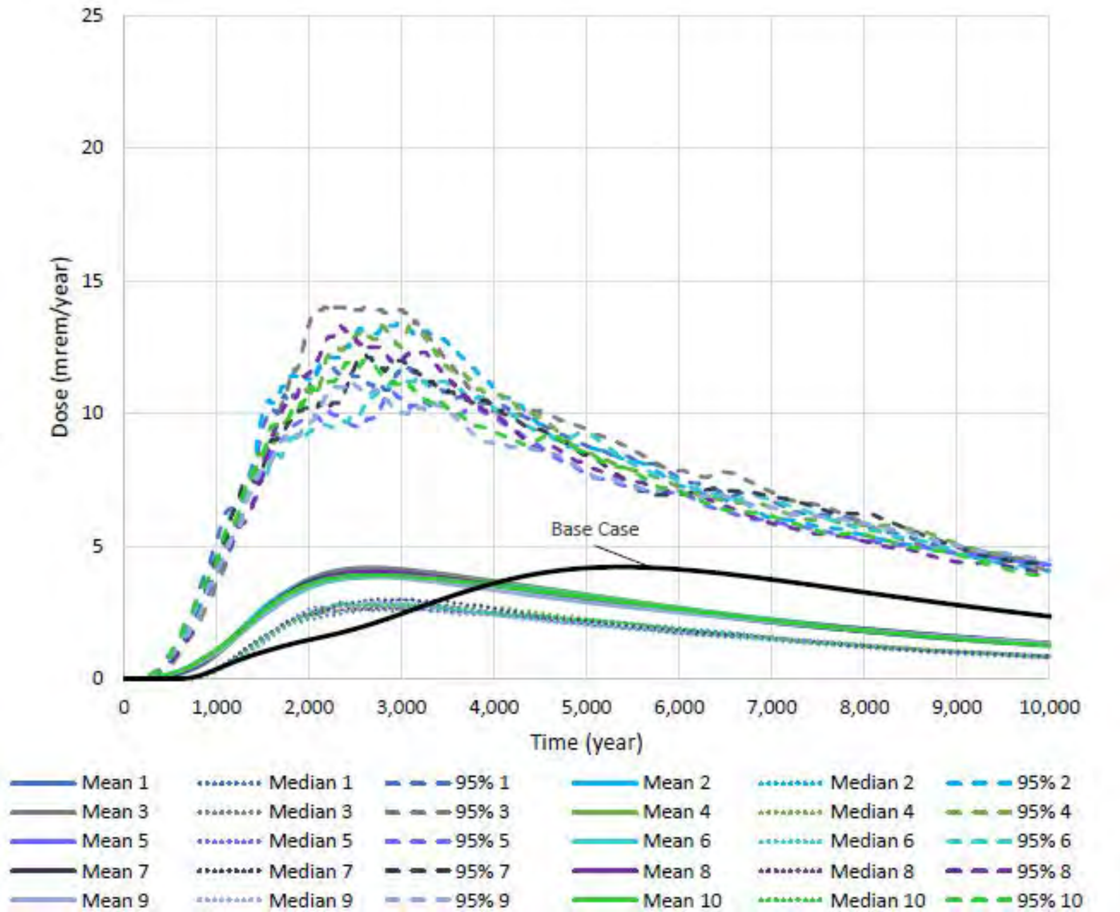

Figure 1. Copy of R1 PA Figure 5.9; Probabilistic results for 10 sets of 300 simulations.

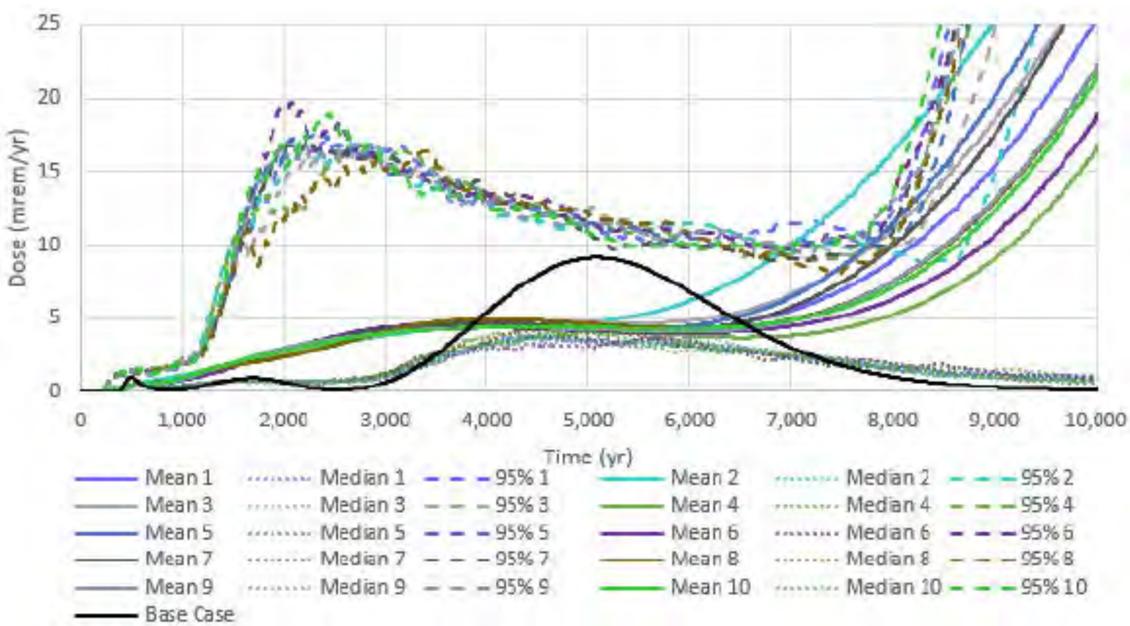

Figure 2. Copy of R2 PA Figure 5.15; Probabilistic results for 10 sets of 300 simulations.

The significant increase in the R2 PA 95th percentile and mean doses occurs because the dose results after ~6K years are skewed high by very large late-in-time doses in some realizations. Section 5.4.2 of the R2 PA states that the divergence of the medians and means "reflects the strong negative[2] skew that develops in the distribution of total dose after 5000 years, due to a large proportion of very small total doses and a small proportion of very high doses." The first peak for the 95th percentiles is attributed in the text to fission products, and the second to actinoids.

Only a handful of radionuclides were evaluated in the probabilistic uncertainty analysis (UA) (C-14, Tc-99, and I-129 through the 1000-year performance period, then adding U-234, U-235, U-238, and Pu-239 for 1000 to 10,000 years). Further, only a subset of RESRAD inputs were evaluated in the UA, and these were selected subjectively to correspond to those perceived to have significant uncertainty, including initial releasable fraction, initial release time, release duration, isotope-specific K_d values, surface runoff coefficient, and precipitation. The scope of the UA was therefore inappropriately constrained. This brings into question the utility of the probabilistic analysis.

As an aside, while modeling to at least 1000 yr is required by the performance objectives outlined in DOE O 435.1, the 10,000 yr duration of these model runs is arbitrary, and clearly does not capture peak dose at any time in the future. The LFRG has made it clear that there is interest in seeing the peak dose in time, no matter when it occurs, in order to provide context for decision-makers. These models should be run long enough to capture peak doses, as is standard practice at other DOE radioactive waste disposal sites.

Appendix C of the R2 PA discusses a variety of system features and processes that could lead to the physical degradation of the engineered cover, which are summarized in Table C.1. As discussed in *DOE Standard, Disposal Authorization Statement and Tank Closure Documentation* DOE-STD-5002-2017 (DOE 2017), Features, Events, and Processes (FEPs, or if exposure scenarios are included, FEPSs) that can affect disposal system performance should be screened for relevance, and those that are applicable should be incorporated into the simulation model used to assess facility performance. Limited screening of some events and processes has been conducted on an *ad hoc* basis, such as the bathtub and sheet erosion evaluations. These evaluations are incomplete and there has been no systematic evaluation of FEPs for inclusion in the EMDF modeling.

2.1.2 PA Critical Issue 2: Bathtubbing Assessment

Critical Issue 2 is a particular example of Critical Issue 1, wherein what appear to be optimistic assumptions regarding minimal and gradual loss of cover performance are invoked to support the assumption that cover infiltration rates will never substantially exceed the rate of water drainage through the landfill liner. In effect, the likelihood of the EMDF being subject to bathtubbing is dismissed based on assumptions layered onto prior assumptions; rather than on a well-supported analysis.

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[2] In this case, the results (i.e., "a large proportion of very small total doses and a small proportion of very high doses") demonstrate a positive skew rather than a negative skew as described in the R2 PA.

In its review of the R1 PA, the LFRG noted (Issue EMDF-S05-PA06-02) that "Analysis of the bathtub scenario should consider the effects of leakage on contaminant transport through the vadose zone to the water table and the resulting groundwater contaminant concentrations and associated doses." Consequently, the bathtub analysis was significantly revised for the R2 document.

Section C.1.1.3 of the R2 PA states, "The composite liner system will include an upper leachate collection system, an underlying leak detection and collection system, and a (*3-ft thick, minimum*) compacted clay leachate barrier" and, "The 5-ft-thick liner system will extend up the sides of the perimeter berms and over the internal berms constructed between disposal cells." The 11-ft-thick cover system includes a (*2-ft thick, minimum*) compacted and amended clay layer overlain by various geomembrane and geotextile layers, drainage, filter and biointrusion layers, and a surface erosion control layer.

To prevent bathtubbing, the rate of infiltration through the cover cannot exceed the rate of drainage through the liner system. As stated in bullet 5 of Section C.1.1.5, the cover must "provide a permeability less than or equal to the permeability of any bottom-liner system or natural subsoil present." As summarized in Table C.1, there are many possible events and processes that can damage cover performance and result in enhanced infiltration (severe storms, floods, landslides, waste subsidence, erosion and gullying, root penetration, thermal and moisture cycles). By contrast, the only events and processes listed that might impact clay liner performance (severe earthquake, improper clay compaction, berm failure) also affect cover performance.

This indicates that the future probability that cover permeability will increase well beyond the rates assumed in the PA from events and processes outside of what the PA quantitatively addresses is likely to be far greater than the probability that liner permeability will increase. This is succinctly expressed in Section C.1.2.2.2 of the R2 PA, "Because liner system clays are more isolated from environmental fluctuations than cover system clay barriers, the liner barriers may retain their safety functions for a longer period."

Scenarios that might result in seepage through berms such as those evaluated in Section C.3 of Appendix C seem plausible. This would only require that infiltration through the cover exceed that through the liner by about one centimeter a year for a couple of centuries sometime during the first millennium after closure of the facility.

With respect to releases due to bathtubbing, R2 PA Section C.3.2 states, "Eventually, saturation and resulting hydrostatic pressure in the waste zone might exceed the confining pressure along a zone of potential weakness (e.g., the seam along the cover/liner interface) and a leachate seep would develop to relieve the pressure. Due to the release of the pressure, catastrophic failure and loss of waste confinement for EMDF would not likely occur."

While the projected impacts to surface water presented in the bathtub scenario in Appendix C appear acceptable when compared to DOE-derived concentration standards, site-specific risk pathways such as fish ingestion were not evaluated. More significantly, the groundwater scenario as analyzed in Appendix C would evidently lead to violations of groundwater protection standards. Note that the leachate concentration for U-238 diluted by two orders of magnitude will yield groundwater concentrations in excess of the 30 μg/L standard for total uranium. Leachate concentrations given in PA Table C-5 for I-129, when similarly diluted, would still be slightly in excess of the 4 millirem per year (mrem/yr) dose standard for beta/gamma emitters, even without consideration of other radionuclides contributing to that dose. Likewise, plutonium concentrations in leachate as given in the table, if no more than a two orders of magnitude dilution/attenuation factor is applied, will drive gross alpha activity greater than the regulatory limit, without consideration of other radionuclides contributing to that activity. The analysis in the PA yields a dilution factor ranging from 99 at 310 years to 14 at 1000 years. Thus, if the bathtub scenario were used to evaluate facility performance, performance objectives stated in section 1.5.1 and specified in section 4.7 of the PA for protection of groundwater resources would not be met.

Rather than using the bathtub scenario to evaluate performance or considering the probability of the failure in performance due to bathtubbing, the PA seems to imply that this analysis provides additional confidence that performance objectives will be met. However, the bathtub analysis is wholly separate from the base case PA. Instead, it should be included with the base case, considering the likelihood that cover degradation will be greater and faster than liner degradation. Using the PA values for leachate concentration and dilution factor, the bathtub scenario in the R2 results in failure of stated performance objectives; i.e., MCLs. Section 3.1 states:

> "Uncertainties in future environmental conditions and the long-term performance of engineered barriers are integrated and generalized in a conceptual model of EMDF performance evolution that is expressed in terms of changes in cover infiltration and leachate release over time (refer to Sect. 3.2.1 and Appendix C, Sect. C.1.3). To address these uncertainties, the PA incorporates a range of potential future conditions defined by selection of input parameter values for model sensitivity evaluations and the uncertainty analysis presented in Sect. 5. In addition, **a separate analysis of the potential impact of an alternative conceptual model of EMDF failure in which cover infiltration greater than liner system release leads to waste saturation and overtopping of the liner (bathtub condition) is provided** in Appendix C, Sect. C.3. 3." [emphasis added]

2.1.3 PA Critical Issue 3: Cover Degradation

Critical Issue 3, like Critical Issue 2, is in part a particular example of Critical Issue 1, where what appear to be optimistic assumptions are invoked regarding minimal and gradual loss of cover performance. It is important to realize that, while the steady-state long-term degraded cover infiltration rate of 0.88 in/yr is based on Hydrologic Evaluation of Landfill Performance (HELP) water balance modeling, the assumptions regarding degraded cover conditions input to HELP are essentially subjective. The maximum assumed rate of infiltration corresponds to roughly one gallon per minute (a small trickle) entering the waste over the entire landfill surface. This scenario for failure of engineered barriers leaves little room for human error in cover design or construction and would seem to require long-term maintenance of the cover.

For evaluation of the EMDF relative to dose-based performance metrics, the as-built condition of the EMDF cover and liner system has been assumed to perfectly reflect the intended design. The intended design is then assumed to exist in perpetuity, with the exception of deterioration of geomembranes; and the small and subjective loss of performance allowed for the cover clay barrier (a two-fold loss of performance) and lateral drainage layer (a three-fold loss of performance). The degradation of performance is assumed to occur in a linear manner between model years 200 and 1000. After model year 1000, degradation is assumed to cease altogether, when accelerated degradation would be expected. Supporting rationale is not provided for the magnitude and rate of performance degradation. In a climate setting that receives 4 to 6 feet of rainfall and on the order of 8 inches of infiltration per year, more rigorous supporting documentation is needed to demonstrate that the EMDF would sit largely unaffected by climate forces and their impacts both on the facility and its setting in the larger Bear Creek valley. Alternatively, uncertainty in long-term cover performance within the PA model should include a quantitative evaluation of the consequences of return to native recharge, as is common practice in other LLW PAs.

These assumptions of minimal cover degradation control the long-term performance of the disposal system to a far greater extent than the model inputs that are varied in the probabilistic uncertainty and sensitivity analyses, and they similarly control the results of the bathtub scenario analysis (see Critical Issue 2). No support is offered to demonstrate how or why degradation of the cover should cease at 1000 years post-closure.

2.1.4 PA Critical Issue 4: Radionuclide Mobility

The R2 PA revises (from the R1 PA) partition coefficients for three radionuclides but fails to address more general concerns with regard to radionuclide release and transport. Because the modeled performance of the disposal system is highly dependent on K_d values, a robust search for applicable K_d data followed by statistical evaluation to develop objective uncertainty distributions is essential. Very little element-specific partition coefficient (K_d) data is presented in the PA, and the statistical distributions used in the probabilistic evaluation of uncertainty are not adequately supported.

The PA acknowledges this to some degree in Section 3.2.2.7:

> To increase confidence in the iodine K_d values applied in the EMDF PA, controls on the partitioning of iodine will be experimentally determined for local site materials (clayey soils and saprolite) derived from the Maryville and Nolichucky Formations. These data will evaluated through the EMDF change control process.

Similar language is present for the discussion of technetium K_d later in the same section of the document. This commitment was first expressed in the LFRG review process in 2018; which raises the question whether these important studies have been initiated.

The K_d values of some key radionuclides were revised for the R2 PA in response to key issues identified by the LFRG review team. The following summary in Table 1 (from slide 6 of DOE-OREM's presentation materials briefing TDEC on PA revisions; 73212_EMDF_PA_05262020.pdf) summarizes the decrease in the R2 PA base case K_d values for three relatively mobile radionuclides that were the primary contributors to base case total dose in the R1 PA:

Table 1. K_d values that were modified for R2.

Decreased partition coefficients for C-14, Tc-99, and I-129
(*higher aqueous concentrations*)

Input Parameter	C-14 K_d (ml/g)	Tc-99 K_d (ml/g)	I-129 K_d (ml/g)
Rev 1 PA K_d (all materials)	1.09	1.0	4
Rev 2 PA waste zone K_d	Zero	0.36	2
Rev 2 PA K_d for non-waste	Zero	0.72	4

The basis of the waste zone K_d values is described in this manner: "Given that approximately one-half of the waste mass is thus similar to saprolite zone material, the K_d values in the waste zone are assumed for the base case to be one-half the K_d values assumed for the saprolite and bedrock zone materials." This approach of layering one assumption onto another assumption is arbitrary and inconsistent with the way chemical partitioning occurs between aqueous and solid phases. If half of the material has one K_d value and the other half has a different K_d value, averaging the two K_d values is improper. Contamination associated with the material with lower K_d will emerge from the waste and show up downstream earlier than that associated with the higher K_d. Averaging the K_d values will not mimic this behavior.

Likewise, proper assignment of partition coefficients to radionuclides present in waste as different chemical species may require modeling desorption from waste for each chemical species separately or a probabilistic approach rather than using an average value. Uranium is likely to be present in waste both as uranium metal and as uranium salts. While metal pieces of uranium will be quite inert, uranium salts and other uranium compounds can be quite soluble and may migrate readily as hexavalent uranium complexed with anions commonly found in groundwater. The K_d values of 50 ml/g assumed for uranium in the PA will not be appropriate for the fraction of uranium disposed in these more mobile forms.

As discussed in R2 PA Section 3.2.2.6, the revised base case K_d values are assumed to be applicable to clay-rich saprolitic and bedrock materials. However, the fundamental assumption of equilibrium partitioning implicit in all the transport models used in the PA is itself not supported by numerous field and laboratory studies. The fractured and weathered saprolite and bedrock of the Maryville and Nolichucky Formations that underlie the proposed facility transmits solute too readily through fractures for the assumed equilibrium partitioning throughout the media to be realized.

As stated in Section 2.1.5.1 of the R2 PA, "Tracer tests and investigations of groundwater contaminant plumes on the ORR (Oak Ridge Reservation) and in BCV (Bear Creek Valley) demonstrate that groundwater tends to move more rapidly along fracture flow paths that are parallel to geologic strike versus flow paths that are perpendicular to strike. This is particularly true for the shallower portions of the saturated zone where most groundwater flux occurs."

In fact, over a dozen field-scale tracer tests, most of which were done with nonreactive tracers and without induced gradients, have been conducted in the Maryville and Nolichucky Formations in Oak Ridge. To summarize these results, saturated zone groundwater velocities in fractures range from a few meters per day to 100 meters per day while migration rates of peak concentrations of nonreactive solutes, retarded significantly by diffusion out of the more permeable fractures, are typically between 10 and 50 meters per year. This diffusion is from concentration gradients between solute in the more permeable fracture networks (the better connected, larger apertures fractures) and that in adjacent portions of the aquifer. In other words, equilibrium conditions were not established in these field scale tracer tests. K_d values derived in laboratory settings where equilibrium between phases may be achieved are not applicable in the in-situ geologic media.

Table 2 summarizes studies performed for in-situ geologic media that report first and peak arrival times for tracers that are considered to be chemically stable in groundwater. These studies were conducted on the Oak Ridge Reservation, under conditions of natural gradient. To the extent possible, field-scale measured retardation of key elements in representative fractured bedrock and saprolite should be applied for this portion of the transport model.

Table 2. Summary of Tracer Testing in Maryville and Nolichuky Formations

tracer test site	geology	reference for geology	tracer	distance (m)	1st arrival (days)	peak arrival (days)	reference for trace
Waste Pit area, Melton Valley	Maryville	Lomenick et al., 1964	tritiated water	3	1	20	Blanco and Parker, 1964
Engineered Test Facility Site, Melton Valley	Nolichucky	Webster, 1996	chlorofluoro-carbons	9	6	60	Vaughan et al., 1982
Engineered Test Facility Site, Melton Valley	Nolichucky	Webster, 1996	tritiated water	9	120	480	Webster, 1996
West Bear Creek Valley CIIDF site	Maryville	Lee et al., 1992	He and Ne gas	35	20	200	McKay et al., 2000
SWSA 5, Melton Valley	Maryville	Jardine et al., 1999	bromide	16.8	3	180	Jardine et al., 1999
SWSA 5, Melton Valley	Maryville	Jardine et al., 1999	He and Ne gas	23	15	180	Sanford et al., 1996
SWSA 5, Melton Valley	Maryville	Jardine et al., 2002	bromide	9	8	100	Jardine et al., 2002
West Bear Creek Valley Well 462 area	Nolichucky	Moline et al., 1998	He gas	15	9	365	Moline and Schreiber, 1996
West Bear Creek Valley Well 462 area	Nolichucky	Moline and Schreiber, 1996	bromide	15	65		Moline and Schreiber, 1996

2.1.5 PA Critical Issue 5: Waste Leaching

For the R2 PA, the RESRAD-OFFSITE radionuclide release mechanism from disposed waste to the environment was changed from "First Order Rate Controlled Release with Transport" to "Instantaneous Equilibrium Desorption." In the R1 PA, uncertainty in the leaching rates of radionuclides was evaluated independently of K_d values, which ignores the strong, non-linear relationship between leaching and K_d for mobile radionuclides shown in Figure 3. During LFRG review conference calls, OREM and UCOR technical staff explained that the uncoupling of K_d values and leach rates was due to RESRAD's inability to calculate leach rate as a function of K_d value when using the "First Order Rate Controlled Release with Transport" release mechanism.

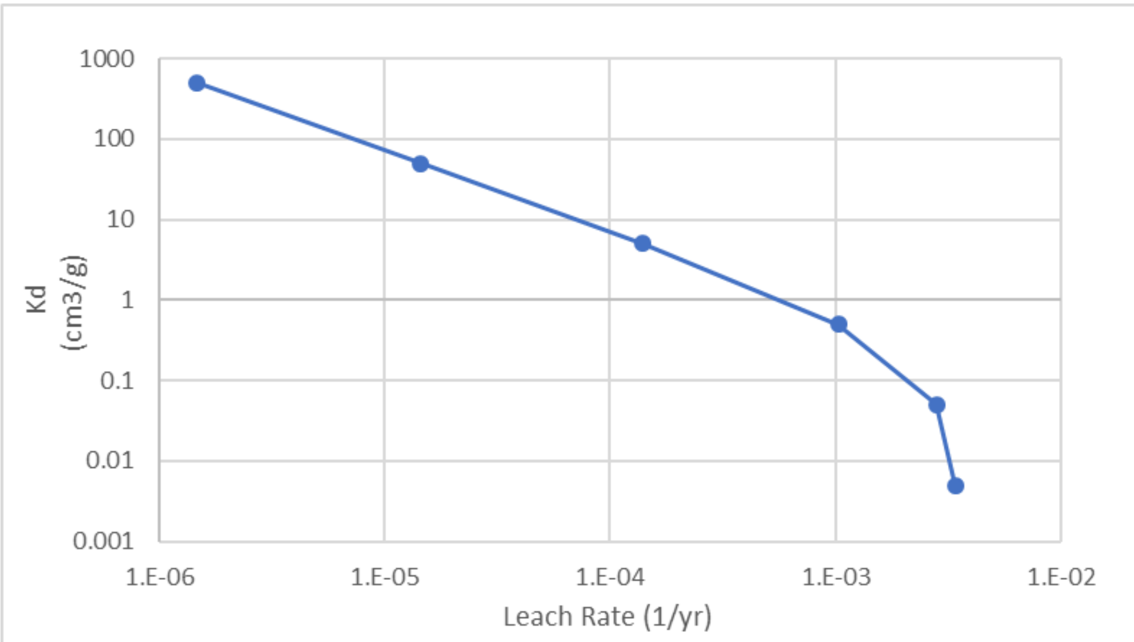

Figure 3. Leach rate as a function of K_d in RESRAD-OFFSITE.

The Instantaneous Equilibrium Desorption release model in RESRAD-OFFSITE allows for internal calculation of leach rates based on K_d values. When applying Instantaneous Equilibrium Desorption release, the R2 PA has adjusted the inventory of four mobile radionuclides to reflect an assumption that 100% of radionuclide inventory is available immediately, leading to leaching (presumably, the leached radionuclides are controlled in a leachate wastewater treatment system) of the following fraction of as-disposed inventories of these radionuclides prior to landfill closure (R2 PA Table G.9):

C-14: 81% of inventory leached prior to landfill closure

H-3: 59% of inventory leached prior to landfill closure

Tc-99: 44% of inventory leached prior to landfill closure

I-129: 14% of inventory leached prior to landfill closure

Assuming Instantaneous Equilibrium Desorption of radionuclide inventory in disposed waste is a protective assumption when applied at closure, but is not protective as applied in the R2 PA. Realistically, disposed inventory of any radionuclide will become available for leaching to the environment based on several factors (i.e., degradation of packaging, degradation of the waste matrix, etc.). Section 3.2.2.5 of the R2 PA states, "The majority of EMDF waste is expected to be disposed in bulk (uncontainerized) form…" but also states, "Radionuclide contamination will include fixed surface contamination as well as contamination distributed within the matrix of more porous materials such as concrete and masonry." This description of waste forms and contamination does not support an assumption of the loss of much of the inventory of C-14, H-3, and Tc-99 during the operational period.

Assuming the loss of much of the inventory of C-14 and Tc-99 during the operational period is important to R2 PA results because, along with I-129, these are the primary drivers of radiological dose during the first 10,000 years post-closure.

Version 3.2 of RESRAD-OFFSITE allows appropriate characterization of radionuclide release under Instantaneous Equilibrium Desorption to account for the effects of engineered barriers, packaging, and waste form. RESRAD-OFFSITE inputs in R2 PA Attachments G.1, G.2, and G.3 show how radionuclide-specific release characteristics have been applied under Instantaneous Equilibrium Desorption to reflect these effects. There are several aspects of the parameterization that appear incorrect, or that are difficult to understand based on the description:

1) The EMDF cover performance model assumes zero infiltration between year 0 and year 200, and then a linear increase in infiltration rate from 0.43 in/yr at year 200 to 0.88 in/yr at year 1000 and in perpetuity thereafter. Section 3.3.4.2 of the R2 PA describes how RESRAD inputs were set to emulate the long-term degraded infiltration rate of 0.88 in/yr. To accommodate the linear transition from 0.43 to 0.88 in/yr over 800 years, Section 3.3.4.2 states, "As a surrogate representation of the assumed increase in cover infiltration over the release duration, … the release model applies a releasable fraction parameter which is increased from zero to one over the 800 year release." This parameterization of the release does not appear to properly represent the intended conceptual model for cover degradation. The releasable fraction should begin at approximately 0.5, not zero, because at year 200 the infiltration rate is approximately 50% of the final (steady-state) value of 0.88 in/yr.

2) The time at which radionuclides become releasable was set at 300 yr for the RESRAD-OFFSITE Base Case, although the base case EMDF performance scenario assumes full design performance (zero infiltration through the cover and into the waste) for a period of only 200 years post-closure. Sections 3.3.4.2 and 3.3.5 of the R2 PA discuss the rationale for adding 100 yr to the release time, based on adjusting RESRAD-OFFSITE waste and vadose zone fluxes to match results modeled in STOMP and MT3D. Adjustment of system-level model inputs to match results of process-level models is appropriate in principle, but the PA only presents such benchmarking of fluxes for Tc-99 (see Section 3.3.5 of the R2 PA). There is no explanation for why the Tc-99 results should be applied globally to all radionuclides regardless of their leachability. Such extrapolation is not intuitive and seems to be contradicted by discussion of relative model performance for less-mobile radionuclides.

3) The adjustments made to the RESRAD-OFFSITE release model parameters for initial releasable fraction and release duration for H-3, C-14, and Tc-99 are discussed in the last paragraph of Section 3.3.4.2 of the R2 PA. The basis of these changes is referenced to Section 3.3.5, but Section 3.3.5 only presents information pertaining to Tc-99, which has a different initial releasable fraction and release duration than H-3 and C-14. Since PA model performance is highly dependent on the RESRAD-OFFSITE release model parameterization of these mobile radionuclides, complete documentation of the basis for adjusting inputs to differ from those consistent with the conceptual model must be provided.

   The conceptual model release profile indicates that the following release parameters should generally apply to all radionuclides in RESRAD-OFFSITE under an assumption of instantaneous equilibrium desorption, unless, like Tc-99, they are adjusted based on radionuclide-specific documented benchmarking with STOMP or MT3D:

   - Linear release of radionuclides over time
   - Release delay time: 200 yr
   - Fraction of radionuclide in the source material that is initially releasable: 0.5
   - Time period for transformation to fully releasable form: 800 yr
   - Total fraction of radionuclide in the source material that is ultimately releasable: 1

2.1.6 PA Critical Issue 6: Inadvertent Human Intruder (IHI) Scenario

The Chronic Post-Drilling scenario provided in Appendix I of the R1 PA addressed exposure only to drill cuttings in a garden. DOE-STD-5002-2017 (DOE 2017) is cited in Section I.3, Appendix I, of the R1 PA as the basis for excluding all groundwater transport pathways. Section 2.2.8 of DOE-STD-5002-2017 states, however, that “The DOE chronic scenario uses a dose measure of 100 mrem/year, but excludes the contributions from drinking contaminated groundwater.” This statement does not require exclusion of all groundwater pathways. Furthermore, DOE-STD-5002-2017 explicitly does not impose requirements, nor does it define the pathways for a Chronic Post-Drilling IHI scenario. There is no logical basis for *excluding* evaluation of groundwater pathways in a Chronic Post-Drilling residential scenario that *includes* exposure to cuttings from a groundwater supply well. Both of these exposure pathways should be included in this exposure scenario.

Neptune conducted a supplemental Resident IHI evaluation using RESRAD-OFFSITE as part of our review of the R1 PA (Appendix B). This evaluation determined that, if infiltration rates exceed the long-term base case assumption of 0.88 in/yr, and if some fraction of the inventory of long-lived and relatively soluble radionuclides have lower K_d values than were assumed for the base case (in particular, substituting a lower value for the K_d of 50 ml/g for U), Resident IHI dose could exceed 25 mrem/yr within a 1000-yr performance period. Doses related to exposure to drill cuttings, which are the only exposures evaluated in the IHI evaluation for both the R1 and R2 PA, were found to be relatively unimportant in comparison to groundwater pathways exposures.

2.1.7 PA Critical Issue 7: Probabilistic Uncertainty and Sensitivity Analyses

The probabilistic uncertainty analysis (UA) and the sensitivity analysis (SA) are incomplete, and many of the parameter input distributions used are poorly documented.

- Section 5.4.1 of the EMDF PA R1 states: "To simplify the analysis and to make total run time shorter, only C-14, Tc-99, and I-129 were included in the probabilistic evaluation for the compliance period." The very limited scope of the UA severely limits its value, and the explanation that the scope of the analysis was constrained because of a desire to simplify and minimize run time is unsupportable given modern computing power. Multivariate probabilistic analysis should not be limited to a pre-selected set of variables. It is particularly important not to restrict distribution development to parameters for which good information is available for developing a distribution. For example, initial radionuclide inventories, and changes to the infiltration rate over time due to cover degradation, should be addressed in the UA to properly understand how the disposal system may perform. This would be more easily solved if a complete probabilistic model was developed and global uncertainty and sensitivity analysis followed.

- The SA presented in the PA is a rudimentary multiple linear regression, based on a handful of preselected input parameters that are evaluated one-at-a-time (OAT). A global SA should be performed as the result of a fully probabilistic PA to assure that the most significant contributors to uncertainty in the results are identified. OAT analyses are incapable of addressing interacting effects, of which there are many in this type of model. This is probably a non-linear model, in which case multiple linear regression is an inappropriate method for evaluation. The EMDF PA is not fully coupled, but it needs to be in order to perform a proper global SA that can vary all inputs simultaneously and use modern statistical (machine learning) methods to find the important (sensitive) inputs to the model.

- Figure ES.13 shows dose result time histories for several SA runs and compares them to the base case results. For most of the first 1000 years, the base case results are lower than the SA cases, implying that the base case is not representative of the system. One would expect that the base case should lie in the middle of the SA results. This reveals bias in the selection of the base case, since it is almost always lower than the SA results during the time shown on this figure.

- The only base case RESRAD-OFFSITE model inputs that were varied in the probabilistic analysis relate to breakthrough times to the aquifer and well concentrations for three radionuclides with relatively small inventories (approximately 8 Ci each of Tc-99 and C-14, and <1 Ci of I-129). At a minimum, the parameter UA should be extended to encompass realistic uncertainty in K_d values for all radionuclides, and to encompass uncertainty in other areas of the model, including inventory and, critically, the infiltration rate. The uncertainty in infiltration rate over time in the UA should encompass more realistic outcomes of long-term cover performance, including infiltration rates approaching those of the native area. Uncertainty in plant and animal transfer factors, surface water body mixing assumptions, and land use and exposure assumptions should also be included in the probabilistic UA. Lastly, because higher infiltration rates lead to more severe bathtubbing, the bathtubbing analysis should be brought into the PA model used to support recommendations.

- The basis of the distributions applied in the UA are inadequately defended. For example, K_d distributions are truncated at a maximum value that is twice the base case value, although sample data exist for K_d measurements of many elements from which to derive data-driven distributions. Distribution development must include review of all available data to create a project dataset, scaling the data as necessary to the time frames and spatial scale of the modeling, and application of statistical methods to create the distribution (Neptune 2015b). In addition, statistical scaling has not been performed when estimating the distributions. It is inappropriate to simply estimate distributions based on available data when the data and the model are presented at different spatial and/or temporal scales. A probabilistic analysis has little value when it is founded on parameter distributions that are not objectively derived from underlying data in a manner amenable to independent review.

2.1.8 PA Critical Issue 8: Water Resource Analysis

DOE Orders 435.1 and 458.1 require a groundwater protection analysis, which amounts to a comparison to Maximum Concentration Limits (MCLs) for radionuclides promulgated by EPA under 40 CFR 141.66 (CFR 2014b) and the corresponding Tennessee Rule 0400-45-01. MCLs include maximum allowable activity concentrations for H-3, Sr-90, Ra-226, Ra-228, and net α (all α emissions other than those from radon and uranium), maximum mass concentrations for total U, and maximum doses from a specific collection of β- and γ-emitters.

For many radionuclides, MCLs correspond to lower concentrations in groundwater than those that would yield a 25 mrem/yr dose. For radionuclides that transport readily via water-borne pathways, performance objectives based on water resource protection are likely to be the most restrictive and to drive waste acceptance limits at the facility. The details of the water resource protection assessment are given in Appendix G (Section G.5.5).

Due to the potential for a large inventory of these contaminants in candidate waste streams, uranium isotopes, particularly uranium-238, and non-radioactive hazardous chemicals, particularly mercury (and uranium as a toxic metal), are likely to pose the greatest threat to water resources from the proposed EMDF. Potential impacts to surface water are not discussed, except in the context of DOE-derived concentration standards, which are based on dose incurred through a water ingestion scenario. Potential impacts on downstream recreational use of surface water are not considered. On p. G-58, the PA states:

> In the absence of local radiological standards for surface water protection, the Derived Concentration Standard (DCS) (DOE 2011c) values are adopted to evaluate impacts to surface water resources.

While there are no local radiological in-stream standards for surface water protection, it should be noted that Tennessee Rule 0400-20-05-.161 does provide limitations on effluent concentrations of radionuclides, which are typically somewhat lower than DOE-derived concentration standards.

The water resource protection assessment indicates that uranium concentrations in groundwater at the point of assessment will be 0.0 μg/L during the compliance period. The MCL for total uranium is 30 μg/L. However, the modeling described in Appendix G predicts that, starting between 20,000 and 25,000 years after closure, uranium will contaminate groundwater above the MCL for at least the next 75,000 years (Figures G.15 and G.16). The advective travel time from the edge of waste to the point of assessment (POA) calculated using the inputs to RESRAD-OFFSITE given in Tables G.10 and G.15 is over 7000 years. Experience with uranium transport from disposal sites in groundwater in Bear Creek Valley and Melton Valley indicates that uranium can certainly migrate through groundwater from sources at the rate of at least one meter per decade, giving a lower bound of 1000 years for the advective travel time to the POA (see also Section 2.1.4). It should be noted that much faster transport is likely, depending on the hydrogeology of the particular site and the chemical form of uranium.

Verification of predicted unsaturated zone transport travel times against field conditions is more difficult. Historical disposal in pits and trenches resulted in sources near the water table, and, in some cases, sources that were seasonally saturated, so there are no ready analogues for the unsaturated zone field conditions assumed at the proposed EMDF site. The PA does not report travel times to the water table explicitly, other than to discuss the results of STOMP modeling in Appendix E. Figures E-26 through E-28 indicate the peak radionuclide flux at the water table is delayed with respect to the peak flux through the liner by about 100, 200, and 1000 years for C-14, Tc-99, and I-129, respectively. Figures E-29 and E-30 indicate a uranium time of travel between the facility liner and the water table of over 10,000 years.

Advective travel times through the unsaturated zone below the geologic buffer can be calculated using values in Table G.14 and input files reproduced in Attachment G.2. This computation yields approximately 60, 275, 1250, and 15,000 years for C-14, Tc-99, I-129, and uranium isotopes, respectively, to traverse the 4.8 m of saprolite underlying the buffer to the water table. Dispersion in the vadose zone is assumed to be small (the Péclet number is approximately 50), so the advective travel time will be a good estimate of the mean travel time given by RESRAD-OFFSITE for these isotopes.

Studies on saprolite in Oak Ridge have concluded that solute transport in the vadose zone is much more rapid when water content approaches saturation and the largest aperture fractures and pores become water filled. Under these conditions, solute concentrations in the vadose zone will vary between the more conductive fracture networks and the less conductive media. Likewise, partitioning of radionuclides between solid and aqueous phases will not reach equilibrium throughout the field of flow.

The long travel times through vadose zone saprolite predicted by STOMP and RESRAD-OFFSITE are the consequence of conceptual model assumptions that limit infiltration through the cover and require that groundwater infiltration and radionuclide release be spatially uniform over the facility footprint. The infiltration rate not only affects the predicted distance to the water table but, via the water content, the unsaturated hydraulic conductivity. The assumption of spatial uniformity over the areal extent of the landfill precludes simulation of discrete failures of barrier layers. Nonuniform barrier failure should be expected and would lead to local increases in relative saturation and more rapid solute transport. STOMP simulations incorporate non-uniformity in saturation at a larger scale due to waste cell geometry, but the sensitivity of the all-pathways model to non-uniform release was not adequately addressed.

Whether or not, as assumed in the PA, "non-uniform release does not result in earlier or larger peak concentrations at the POA", it is clear that more realistic modeling of vadose zone solute transport would result in larger concentrations of radionuclides such as Tc-99, I-129, and uranium at the POA within the compliance period. Given that the RESRAD-OFFSITE model has been adjusted to agree with the STOMP results, that STOMP fails to account for fracture flow in the residuum, and that the partition coefficient assumed for uranium in both models is unrealistically high for some uranium in some chemical states, uranium seems likely to reach both the water table and the POA much sooner than indicated by the PA.

The models used to simulate contaminant transport do conserve mass and do not account for any irreversible chemical reactions of dissolved uranium with minerals. Thus, at least the total predicted flux of uranium over time reaching the POA might be conservative. The model appears to overestimate the travel time of uranium through the subsurface by millennia, and the predicted concentrations would indicate contamination of groundwater for tens of thousands of years.

If potential impacts to water resources are to be realistically evaluated, a modeling strategy that yields results more consistent with experience in field situations at Oak Ridge is needed. Likewise, an assessment that includes at least some consideration of impacts to water resources due to toxicity effects of uranium and mercury would is preferable, as these impacts could be significant enough as to potentially limit landfill operations and waste acceptance at the facility.

2.1.9 PA Critical Issue 9: Radon Ground Surface Flux Analysis

One of the performance metrics for both DOE O 435.1 and 458.1 is that the average flux of radon at the ground surface be maintained below 0.74 $Bq/m^2 \cdot s$ (20 $pCi/m^2 \cdot s$). Appendix H in the PA analyzes performance against this metric. This analysis applies a deprecated methodology for calculating radon flux, and it uses default rather than site-specific input parameters. The analysis does not account for migration of radon parents into the cover, thereby potentially underpredicting the ground surface flux.

The calculation is based on an old reference, *Radon Attenuation Handbook for Uranium Mill Tailings Cover Design* (Rogers et al. 1984), which has been updated and improved by its surviving author, Kirk Nielson, and his colleague Gary Sandquist (Nielson and Sandquist 2011). The 1984 analysis was developed for the narrow application to uranium mill tailings piles consisting of a uniform waste form of uranium ore tailings, and a uniform cover of a single material, and "…did not consider any surface radon flux contribution from radium in the covers" (Rogers et al. 1984). While a semi-analytical approach is outlined for multilayered covers, this was later refined in follow-up work by Rogers and Nielson (1988), also developed in NRC (1989). Even the 1989 approach, however, failed to account for the potentially significant factor of radon parents (notably Ra-226) migrating into the cover due to water phase diffusion and biointrusion. Both the 1984 and 1989 approaches fail to include this coupled transport mechanism. Furthermore, the 1984 default parameters were used (PA Appendix H, Section H.1); these should be replaced with site-specific values where available.

Neptune worked with Nielson and Sandquist to develop a robust methodology for the modeling of radon ground surface flux (Neptune 2015e; Nielson and Sandquist 2011). It was determined that the only way to adequately model radon flux was to use a mechanistic gaseous diffusion model that was coupled with the processes of radioactive decay and ingrowth, water phase diffusion of radon and its parents, and biotically induced transport of radon parents into the cover. There is no analytical solution for this. This methodology is documented in Neptune (2015e).

In addition to using an outdated approach for modeling radon flux, some of the input parameters used in the EMDF PA modeling will lead to underprediction of the radon flux. The fraction of Rn-222 produced by decay of Ra-226 that is released from the solid matrix is known as the escape-to-production ratio (E/P ratio) as well as the emanation coefficient, the emanation factor, or emanating power (Nielson and Sandquist 2011). The E/P ratio describes that fraction of Rn-222 that stops in the air or water-filled pore space and is free to diffuse. For cover materials that were initially uncontaminated but host Ra-226 (and potentially other parents in the uranium series), the radium is present in interstitial waters or on grain surfaces. Because the radium is not adsorbed deeply into the matrix, its decay will generally produce radon in the air- and water-filled pore spaces in the cover material. This would correspond to an E/P ratio for materials other than waste of 1 (Nielson and Sandquist 2011). The PA's choice of 0.25 (R2 PA Table H.4) will lead to underprediction of the radon flux. This choice is made based on values reported for soils in, for example, the RESRAD documentation. But that applies to soils that have radon parents in their grain matrices, not soils that are merely surface-contaminated.

The free-air diffusion coefficient for radon gas should be 0.11 cm^2/s (Rogers and Nielson 1991). The R2 PA does not document the value used in the Appendix H analysis.

Long-term estimates of radon surface flux are dependent on assumptions regarding the rate and type of cover erosion. The EMDF PA radon evaluation should evaluate the impact of uncertainty in long-term erosion rates on radon surface flux.

The calculation of radon flux in the R2 PA is limited to an example at 1000 yr following site closure. The amount of Ra-226 in the system, however, would be expected to increase substantially due to the long-lived parents (e.g., U-238). The radon flux calculation needs to be evaluated through time in order to determine its actual peak.

2.2 Key Findings—Performance Assessment

A discussion of the most significant findings of this review is provided here. Key findings are offered to provide additional context regarding the review. Sections 2.2.1 through 2.2.6 are related to critical issues 1 through 6. Other key findings are listed and described subsequently.

2.2.1 Conceptual Model Assumptions

The following observations apply to the three key parameter assumptions. Indented text is quoted from the PA.

With respect to the three key parameter assumptions for EMDF compliance:

1) Iodine-129 K_d values for the engineered barriers and geologic materials below the EMDF liner are modeled to be greater than 1 cm^3/g.

As discussed in R2 PA Section 3.2.2.6, the revised base case K_d values are applicable to clay-rich saprolitic and bedrock materials. This assumption is not defensible in the fractured and weathered bedrock that underlie the proposed site, as flow through fractures in these media has been demonstrated in most cases to be too fast to allow equilibrium partitioning of solute between phases to be achieved.

2) IF the I-129 K_d value is less than 1.5 cm^3/g, THEN: the values for cover infiltration, vadose zone thickness, and saturated zone flux (Darcy velocity) must satisfy certain conditions.

As noted for observation 1 above, there is good reason to conclude the I-129 retardation in fractured bedrock will be much less than that implied by a K_d value of 1.5 cm^3/g in soil or sediment.

3) The estimated post-closure EMDF average I-129 activity concentration is less than 0.41 pCi/g.

The base case post-closure waste concentration is 0.35 pCi/g, which provides very little margin. Per Table 3.3 of the R2 PA, the as-generated waste is estimated to have an I-129 concentration of 0.766 pCi/g, and an as-disposed concentration of 0.407 pCi/g. The reduction of I-129 inventory from the as-disposed concentration to the post-closure concentration by 14% is due to assumed operational losses and reductions in mobility resulting from treatment of collected leachate during active facility operations (pre-closure). The R2 PA (p. 138, Section 3.2.2.5) claims that "Taking credit for operational period losses is conceptually consistent with the equilibrium desorption model for radionuclide release adopted for the PA models." But the RESRAD equilibrium desorption release model is specifically structured to allow the desorption model for each radionuclide to be applied at user-defined time(s) when the radionuclide is assumed to become available for release. The PA assumes that all soluble radionuclides will be disposed in a manner that renders 100% of their activity to be immediately available. This is not protective, lacking supporting documentation regarding the waste form. Effectively, the pre-closure EMDF facility is modeled to function in practice as a waste treatment system. Note also that the assumed inventory reductions of C-14 (>80%), H-3 (60%), and Tc-99 (55%) are more aggressive than for I-129. This is potentially significant, as C-14, Tc-99, and I-129 are the main contributors to base case dose.

Taken together, the three key parameter assumptions represent conditions necessary to ensure sufficient delay of iodine reaching the POA or sufficient dilution upon reaching the POA. However, travel times of bromide determined by tracer tests in Conasauga Group saprolite and shallow bedrock (Table 2 above) indicate that travel times of halides from a release at the facility to the points of assessment will be on the order of years to decades.

Both laboratory and field solute transport experiments conducted at Oak Ridge indicate that impacts of I-129 will almost certainly be felt much more quickly than the PA predicts. Given the rapid migration through groundwater or surface water, the timing of impacts at the POAs for groundwater and surface water will depend primarily on the timing of release from the engineered facility. Reduction of the assumed partition coefficient for I-129 in the liner and buffer by a factor of two and correction of travel times in saprolite and shallow bedrock to values based on measured migration rates of bromide would result in significantly higher contributions of I-129 to potential dose delivered to a receptor at times less than 1000 years post-closure.

It is not clear how the laboratory measurements discussed in detail in Section 3.2.2.7 of the R2 PA will resolve uncertainties concerning migration velocities for iodine in geologic materials below and downgradient of the EMDF. While laboratory determination of a partition coefficient might be useful for estimating leachate concentrations and retardation of iodine migration through engineered barriers, abundant evidence from studies in Oak Ridge suggests that the retardation in the geologic media over a distance of 100 meters or more will be determined primarily by the properties of the fractured bedrock and the diffusion coefficient. Some adsorption on fracture surfaces will occur, but investigations of transport in Conasauga Group rocks using models that incorporate the physics of transport in discrete fractures reveal that the assumed distribution coefficient for halides and other mobile chemical species does not contribute significantly to retardation in the saturated zone.

Key assumptions that are not explicitly acknowledged:

- The approach to modeling the transport of contaminants in groundwater, critical to the evaluation of water resource protection and dose in the all-pathways exposure scenario, relies on assumptions that are not explicitly identified in the PA as key assumptions. These assumptions lead to significantly longer travel times to a point of assessment for many of the radionuclides considered in the PA than can be justified by decades of tracer studies and plume monitoring data in Oak Ridge. Section 1.3 of the R2 PA states:

  > Remedial investigation of historical waste disposal sites in BCV and elsewhere on the ORR and ongoing CERCLA remedial effectiveness monitoring (DOE 2017c) have provided extensive insight into the likely behavior of the EMDF system in the decades following closure, once the performance of engineered systems begins to degrade.

  The PA summarizes much information derived from Oak Ridge studies but fails to properly incorporate that information into the modeling effort. In fact, the approach to modeling transport of contaminants adopted in the PA was rejected by the team preparing the remedial investigation (RI) report for Bear Creek Valley in the 1990s. The strategy for modeling contaminant transport in the RI is explained on page E4-1 of that document (DOE, 1997):

  > The complex nature of the hydrogeology and contaminants in BCV preclude development of a single numerical computer model to describe fate and transport of contaminants at this site. Rather, a combination of small-scale numerical transport models, an analytical groundwater transport model, a geochemical model, and simple estimates of contaminant attenuation/dilution along specific pathways are combined in the framework of the conceptual model for fate and transport analysis.

  Rather than base transport modeling on a porous media groundwater flow model coupled with the assumption of equilibrium partitioning of solutes between solid and aqueous phases throughout the flow domain, it would be better to choose a model that incorporates flow through discrete fractures to predict solute transport.

  Groundwater flow in Bear Creek Valley is known to be dominated by preferential flow through fractures and dissolution features. The PA fails to use the decades of studies and monitoring that occurred during active waste disposal in Melton Valley as a reality check on modeling results. The early efforts to understand and better manage the release and migration of radionuclides in the Conasauga Group formations were documented in Health Physics Division reports throughout the 1950s and 1960s. The information in these reports, along with results from more recent studies, has been summarized in a number of publications (e.g., Manneschmidt and Witkowski, 1967, Webster, 1976, Olsen and others, 1986, Spalding, 1987, Huff, 2000).

The well-documented migration of many waste constituents disposed in various forms in Conasauga Group rocks at Oak Ridge indicates that many radionuclides can be transported 100 meters through groundwater in a few decades after reaching the water table in this hydrogeologic setting. The inability of the geologic formations in Oak Ridge to retard contaminant migration led to the evolution of waste disposal practices from unlined pits and trenches during the 1950s and 1960s to hydrofracture in the 1960s to the 1980s, to lined disposal units during the 1990s. This evolution culminated in the use of tumulus facilities for on-site disposal and, finally, in the late 1990s, on-site disposal of radioactive waste generated by non-CERCLA operations in Oak Ridge was discontinued.

- FEPs that influence system performance are typically examined and catalogued as part of performance assessment and determination of the safety case for radiological waste disposal sites (Andersson et al. 1989; Bechtel SAIC 2002, 2005; Burkholder 1979; Cranwell et al. 1982; Freeze 2012; Guzowski 1991; Guzowski and Newman 1993; Hertzler and Atwood 1989; Hommel 2012a, 2012b; IAEA 1983, 1985, 2012; Koplik et al. 1982; NEA 2000; Neptune 2015a, 2015c, 2015d, 2016; NRC 2003, 2013; Price et al. 2007; Seitz 2014; Shipers 1989; SNL 2008; Tauxe 2012; USACE 2012). FEPs analyses should be among the first items in the PA development workflow, and Section 3.2 and Appendix C, Section C.1.2 of the PA mentions FEPs: "The features, events, and processes identified provide the basis for a conceptual model of EMDF performance evolution that generalizes performance in terms of cover infiltration and leachate production." Examples of significant FEPs that should be evaluated include significant erosion events and processes, bathtubbing, substantial failure of the performance of the lateral drainage layer and/or clay barrier layer, and future occupation of the site.

- Sheet and gully erosion models should be developed and applied in a probabilistic manner to the PA and CA. Alternatives to the RUSLE2 agricultural erosion model used to evaluate erosion in Appendix C, Section C.4 should be developed, because of the presumed future (naturalized) state of the landfill described in Section C.1.2.1: "…long-term evolution of the cover surface soil will be constrained by the local climate and ecological processes that govern the succession of biological communities over time. In particular, it is likely that once the site is no longer actively maintained, it will eventually become forested." In addition, RUSLE2 is not suited to modeling a forested environment with steep slopes.

  Several studies support the conclusion that events and processes such as gullying and hillslope failure will eventually undermine any engineered cover design such as the EMDF (with steep slopes flanked by active surface water features), leaving wastes exposed at the ground surface (Alonso et al. 2002; Bennett 1999; Bennett and Casalí 2001; Boothroyd et al. 1979; Hancock et al. 2014; McKinney 1986; Poesen et al. 2011; Shipers 1989; Smith and Benson 2016; Smith et al. 1997; Tucker and Doty 2018; Waugh and Richardson 1997; Willgoose and Hancock 2011). The EMDF PA RESRAD base case model assumption of a zero-erosion rate is not defensible.

Long-term erosion rates should be evaluated by use of analog sites and study of regional landscape evolution (Section 3.5.3 of NUREG-1757, Volume 2, NRC (2006)). For example, recent applications of LiDAR in archeology have allowed evaluation of the evolution of natural and modified landscapes in hilly and forested terrain that could be applicable to the assessment of long-term performance of engineered disposal systems such as the EMDF. Landscape evolution models, such as the LandLab model, supported by terrain and age dating analyses, could be used to evaluate potential long-term risks posed by Bear Creek and Pine Ridge tributary erosional processes, as well as development of hillside gullies on the cover.

- The R2 PA, Appendix C, Section C.1.2.2.1 states: "Another process that can compromise the function of cover components is post-closure differential settlement (subsidence) of the waste. … Due to the variety and heterogeneous nature of expected EMDF waste forms and the resulting potential for subsidence that could impair cover system functions, this degradation mechanism is an important uncertainty in the conceptual model of EMDF performance evolution." If the impacts of subsidence are potentially significant, this degradation mechanism should be addressed in the PA modeling (e.g., by assuming localized thinning of the waste and cover layers and enhanced infiltration). This important uncertainty was not addressed in the R2 PA.

- Biotically induced contaminant transport, such as plant root uptake, is omitted from the modeling with incomplete justification. The R1 PA cites Jackson et al. (1996), stating that "typically more than 75 percent of temperate deciduous forest root systems are limited to the upper 50 cm of the soil profile." This is not a reference that is local to East Tennessee, and it also ignores potentially significant uptake from deeper roots. There are no references provided in the PA for root mass distribution with depth for local forests, though these could likely be obtained from the University of Tennessee Arboretum, which is located near the Central Bear Creek Valley site. It has been shown that biotically induced transport can be significant in bringing radionuclides to the ground surface (Shott et al. 2000a). Historical research conducted at Oak Ridge National Laboratory (ORNL) further validates this (Auerbach 1993), corroborating work done at other sites (Breuer et al. 2003; Crow 1978; Hoven et al. 2000; Leithead et al. 1971; SWCA 2011; Waugh and Richardson 1997; Whittaker and Woodwell 1968). The R2 PA also does not provide justification for excluding this important contaminant transport pathway, which should have been considered in a FEPs analysis.

  The PA assumes that the low moisture retention of the very coarse material of the biointrusion layer will limit root growth until sufficient fine materials have accumulated. This assumption is offered as justification for omitting biotically induced contaminant transport from the modeling (R2 PA Appendix C, Section C.1.2). Such statements require support. It is also possible that, in times of water stress, roots could be found to traverse such layers in search of more water at depth.

  In consideration of burrowing animals, it is acknowledged that a thick layer of cobbles (defined as having diameters larger than 80 mm) is likely to deter small burrowing mammals (Blatt et al. 1972). However, there is no such effect on larger fossorial mammals (e.g., badgers and foxes) or on invertebrates such as ants that favor nest-building in open pore spaces. Research is available to help parameterize animal contaminant transport activity (Gonzales et al. 1995; Leithead et al. 1971; MacKay 1993; SWCA 2011, 2012). This potentially important contaminant transport pathway was not considered in the PA.

- The effect of climate change on precipitation and subsequent infiltration into and erosion of the EMDF cover system has not been adequately addressed. Beyond a cursory sensitivity analysis to evaluate the uncertainty in long-term cover performance using a factor of 1.25 on the precipitation rate (Appendix G, Section G.6.2.3), no further consideration was given to the potential impacts of climate change.

  Section 5.3 of the R2 PA includes a discussion of a revised single-factor RESRAD-OFFSITE sensitivity analysis performed to assess the impact of future changes in precipitation and runoff on infiltration and dose. In the R2 PA, sensitivity analysis assumptions regarding runoff led to a range of cover infiltration from 0.43 to 4.0 in/yr. Note that 4.0 in/yr is about half of the present-day infiltration rate of 8 in/yr for the native surrounding area. The text of Section 5.3 states:

  > The RESRAD-OFFSITE release model (instantaneous equilibrium release option) and one-dimensional vadose zone representation appear to over-predict the activity flux from EMDF for radionuclides having Kd values > 1 cm^3/g, including I-129 and U-234 (refer to Sect. 3.3.5 and Appendix G, Sect. G.5.6). The sensitivity evaluation on the lower runoff coefficient value (0.83) corresponding to 4 in/yr cover infiltration produced extremely large doses after 5000 years that are associated with actinides (e.g., U-234 and Pu-239) in the EMDF estimated inventory. These extreme dose levels are not likely representative of future releases of uranium and plutonium for EMDF, and so the results of the sensitivity evaluation for the runoff coefficient are presented only for the total dose associated with C-14, Tc-99 and I-129 in Fig. 5.10.

  This illustrates the fundamental concern with this PA described in Critical Issue 1; i.e., that the behavior of the system is controlled by assumptions rather than revealed by modeling. Section 5.1 of the R2 PA addresses sensitivity and uncertainty analyses using STOMP, including the effect of increased vadose zone infiltration, but the analyses are limited to Tc-99 and only a two-fold increase in the base case post 1000-year infiltration rate of 0.88 in/yr. The scope of the sensitivity and uncertainty analyses was narrowly constrained, with the result being that the full magnitude of potential future doses is not shown.

  Section 7.2.1 of the R2 PA acknowledges that "Uncertainty in future annual average precipitation and the degree of cover system degradation (two fundamental controls on cover infiltration) are two of the key parameter uncertainties identified in the RESRAD-OFFSITE probabilistic uncertainty analysis." Section 7.2.1 further states that "In general, the earthen cover components overlying the HDPE and clay infiltration barriers should be relatively stable under the natural range of environmental conditions, even considering natural climate fluctuations or the potential for progressive climate change." No basis is offered for this statement.

  An average annual total precipitation of 54.39 inches was used as an input for the HELP model, but erosion is caused more by intense, single storm events than average precipitation. This average is based on the daily precipitation data for Oak Ridge, Tennessee, from 1961–

1990. HELP modeling did not consider higher maximum annual precipitation amounts recorded during a similar period (83.9 inches from 1948 through 2015, according to the National Climatic Data Center) or higher potential precipitation resulting from climate change.

Furthermore, climate change may contribute to degradation in system performance in ways other than increased annual-average precipitation. For example, an increased frequency of high-intensity rainfall events could exacerbate cover erosion, and climate-related changes to precipitation and plant communities for a naturalized cover could affect future evapotranspiration potential. The potential impacts of climate change should be evaluated using a fully coupled model of system performance to understand the sensitivity of model results to climate-related phenomena.

- Loss of institutional control (IC). The assumption of perpetual IC is optimistic, not pessimistic, since there is no evaluation of an on-site resident. While IC that is maintained for more than 100 years would be expected to limit direct exposure, ongoing monitoring and maintenance would be needed to ensure protection of water resources.

2.2.2 Bathtubbing Assessment

- The bathtub analysis did not evaluate the ingrowth of short-lived radionuclides in decay series during the modeling period. This resulted in calculated future concentrations of several radionuclides being negligible when in fact future concentrations can be expected to be greater than current levels due to ingrowth from long-lived parents. These radionuclides include many not considered in the PA and CA, such as the following decay products from U-235 and U-238 with half-lives over 5 yr: Ac-227, Pa-231, Ra-226, Ra-228, Rn-222, Th-228, Th-229, Th-230, and U-234. The doses from these could be significant, and their analysis should be included in a comprehensive PA.

- In R2 PA Section 5.3, the sensitivity analysis evaluated cover infiltration up to a value of 4.0 in/yr, which is about one-half of the 8 in/yr infiltration rate of the native surrounding area. If the seepage rate is linearly related to infiltration rate, a 4.0 in/yr rate would produce seepage of 5.0 gpm. The PA should provide information regarding whether such a seepage rate could present a risk of erosion and/or undermining.

2.2.3 Cover Degradation

- The PA does not provide a basis for the rate of clay layer degradation (represented only by a change in saturated hydraulic conductivity [K_{sat}] of 3.5E-08 cm/s to 7.0E-08 cm/s over 800 years) and lateral drainage degradation (a change in K_{sat} of 0.3 cm/s to 0.1 cm/s over 800 years), nor an explanation for why degradation ceases at model year 1000. Further, a change in K_{sat} implies a change in other porous medium physical properties that are closely related to K_{sat}, such as effective (hydraulic) porosity, but such relationships are not considered in the modeling.

This representation of performance degradation over time appears to be arbitrary. The narrow changes in modeled performance are at odds with statements in the PA regarding state-of-knowledge:

> Although a general progression from full design performance to some long-term degraded performance condition will occur, the timing and magnitude of degradation is quite uncertain, particularly given the potential interactions among the various disposal system elements, safety functions, and degradation processes described above," and, "Eventually, severe weather events and progressive climate and vegetation changes can lead to erosion of the protective cover components and accelerate degradation of the clay barrier in the cover, increasing the likelihood of greater water infiltration over time. (Appendix C, Section C.1.2.2.2)
>
> EMDF performance is expressed in terms of changes in cover infiltration and leachate release, beginning at the time of final cap completion and facility closure. (Section C.1.3)

Eventually, this naturalization enters "a final period during which water flux into and out of the disposal unit reaches some long-term, relatively stable limit." That naturalization may progress until cover infiltration reaches the rate of natural recharge is also indicated in the generalized conceptual model of the EMDF (Figure C.4).

- Leaching based on a future, long-term infiltration rate approaching that of the surrounding natural environment should be modeled. This eventuality is noted as a possibility in PA Appendix C, but the analysis seems not to have been documented.

- Supplemental RESRAD modeling by Neptune as part of the R1 PA review (Appendix B) indicates that if a naturalized infiltration rate of 8 in/yr is used in the modeling, and if some fraction of the inventory of uranium and long-lived soluble radionuclides has lower K_d values (using RESRAD Data Collection Handbook, or DCH, values) than those assumed for the base case, doses could exceed the 25 mrem/yr threshold at about 1,100 years.

- Neptune's supplemental RESRAD modeling also indicates that the assumed failure time of the cover geomembrane does not influence the shape of the release and dose curves; it simply controls the "start time" of when long-lived and mobile radionuclides (C-14 and Tc-99) are leached. Delaying release of long-lived mobile radionuclides is not necessarily desirable, since delaying the time when mobile radionuclides are leached increases the probability that effective institutional controls (monitoring and interruption of potentially complete exposure pathways) will be lost by the time these releases occur.

2.2.4 Radionuclide Mobility

- The PA and CA evaluate risks from only a small handful of constituents proposed for disposal: H-3, C-14, Tc-99, and I-129. Many other potential contributors to radiological dose and risk have been practically omitted from the analysis due to a combination of relatively large assumed K_d values and an assumption of negligible degradation of the performance of the engineered cover over both 1000- and 10,000-year periods of assessment. The most significant omission from the analysis is evaluation of relatively mobile forms of uranium, and its various isotopes and progeny.

- In dismissing other radionuclides from the analysis, their progeny are dismissed as well. Some of these progeny might have a low retardation factor and high dose effects, and should be considered. For example, the decay chain of U-238, even when limited to progeny with half-lives over 5 years, includes U-234, Th-230, Ra-226, Rn-222, and Pb-210. Radon-222 is a noble gas (with zero retardation) and although omitted from dose analyses in air, it can contribute strongly to doses by other exposure pathways and deposits another strong dose contributor, Pb-210 (and its progeny), in locations near the ground surface. Once U-238 progeny achieve secular equilibrium, doses from what was once purified U-238 can increase by orders of magnitude. The issue of the exclusion of doses from progeny (and specifically external doses from radon progeny) is not addressed in the R2 PA.

- The PA models a variety of materials using the same K_d values, which is not in keeping with common practice. Even the older Baes et al. (1984) and Sheppard and Thibault (1990) references provide different values for different materials. Approximately 50% of the waste is expected to consist of debris with characteristics very different from those of local soil. Critically, this statement (R2 PA Executive Summary, p. ES-10) may not be correct: "Under a long-term performance scenario, contaminant retardation in the vadose zone beneath EMDF and within the saturated matrix of the fractured rock at the CBCV (Central Bear Creek Valley) site serve disposal system safety functions by delaying and attenuating impacts of radionuclide release at potential groundwater and surface water exposure points." Retardation is reduced in the fracture-dominated flow of the saturated zone. By applying the same K_d values in the fractured rock zone as at other locations in the model domain, long-term performance is overestimated. Accordingly, this approach understates long-term contaminant transport and dose consequences.

- Neptune's supplemental RESRAD modeling indicates that near-term (<1000 years) and long-term performance is substantially poorer than that shown in the PA when substituting recommended K_d values (geometric mean) for clay soil type (most analogous to shale) from the RESRAD DCH, Table 2.13.3 (Yu et al. 2015) for the base case values used in the PA. Base case K_d values are lower (more "conservative" inasmuch as contaminants move more quickly via water pathways) than the RESRAD DCH K_d values for elements with relatively large K_d values. However, the opposite is true for uranium and the more-soluble elements hydrogen, carbon, and technetium, and it is these more-soluble elements that are responsible for water-pathways doses. The influence on modeled future doses from using these lower K_d values is particularly evident if infiltration rates exceed the 1 in/yr "degraded condition" value assumed in the PA.

2.2.5 Inadvertent Human Intruder (IHI) Scenario

- The basis for the distinction made in PA Appendix I between IHI scenarios is not clearly described. Section 1.6 of the R2 PA states that "BCV (Bear Creek Valley) will remain under DOE control and within DOE ORR boundaries for the foreseeable future." But the qualification of "for the foreseeable future" can be interpreted to be, practically, the entire 10,000-year modeling period. Institutional controls are unlikely to be effective for this timeframe (Applegate and Dycus 1998; NRC 2011). The PA should explore cases where IC is lost shortly after closure.

- At the time of loss of effective institutional control, the point of assessment (POA) for a member of the public (MOP) should have no artificial restriction, such as the 100-meter boundary used in the analysis. DOE M 435.1 suggests, but does not require, the use of this 100-meter boundary. The DOE Manual allows for a larger or smaller buffer zone to allow for site-specific conditions. Insofar as a 100-meter buffer, or any buffer, is established and enforced only by institutional control, the use of such a buffer throughout the modeling period effectively assumes perpetual institutional control during this period.

- DOE-STD-5002-2017 (DOE 2017) is cited in PA R2 Appendix I, Section I.3, as the basis for excluding all groundwater transport pathways from the Chronic Post-Drilling IHI scenario described in Appendix I. However, DOE-STD-5002-2017 does not specifically define the pathways for a Chronic Post-Drilling IHI scenario. Section 2.2.8 of DOE-STD-5002-2017 alludes to a DOE scenario when stating, "The DOE chronic scenario uses a dose measure of 100 mrem/year, but excludes the contributions from drinking contaminated groundwater." This statement does not support exclusion of all groundwater pathways, and DOE-STD-5002-2017 does not elsewhere describe the attributes of a DOE chronic IHI scenario. Critically, Section 2 of DOE-STD-5002-2017 provides guidance for conducting a PA and explicitly states that it does not impose requirements. No logical basis is provided in the EMDF PA for excluding evaluation of groundwater pathways, and other potentially applicable pathways such as external irradiation in a basement, from a Chronic Post-Drilling residential IHI scenario. Regardless of whether groundwater is expected to be used as a potable supply, groundwater protection must evaluate radionuclide concentrations with respect to the MCLs, as discussed above.

- Neptune's supplemental RESRAD modeling (Appendix B) was conducted for a Chronic Post-Drilling IHI scenario, where this scenario included all realistic groundwater exposure pathways in addition to drill cuttings. The supplemental Resident IHI evaluation used a model constructed with the RESRAD-ONSITE computer program to assess dose for a hypothetical resident located on the EMDF landfill, who utilizes a domestic groundwater well for home, garden, and watering small domestic livestock (poultry). The impact of uncertainty in K_d values, long-term infiltration rate, and time of geomembrane failure were evaluated in the Resident IHI supplemental evaluation. Leach rates were calculated internally in the RESRAD program, rather than being defined independently by the modeler, as was done in the R1 PA.

  The supplemental Resident IHI evaluation determined that, if infiltration rates exceed the long-term base case assumption of 1 in/year, and if some fraction of the inventory of uranium and long-lived soluble radionuclides have lower K_d values (default values from the RESRAD program documentation were applied) than were assumed for the base case, Resident IHI dose could exceed the 25 mrem/yr performance metric within a 1000-yr performance period. Doses in excess of 1000 mrem/yr were modeled using RESRAD default K_d values at 3000 years post-closure. Dose related to exposure to drill cuttings, which were the only exposures evaluated in the EMDF PA IHI evaluation, were found to be relatively unimportant in comparison to groundwater pathways exposures.

2.2.6 Conceptual Model for Groundwater Pathway

The basic "modelability" of the setting of the EMDF for near-surface disposal of radioactive waste has not been adequately demonstrated. Although it is a siting requirement of the Nuclear Regulatory Commission rather than that of DOE, 10 CFR 61.50, *Licensing Requirements for Land Disposal of Radioactive Waste* (CFR 2014a), describes the attributes of site suitability for near-surface disposal. A fundamental property of a suitable site is that, "The disposal site shall be capable of being characterized, modeled, analyzed and monitored." The ORR, with its steeply inclined geologic strata and abundant karst features that provide fast pathways for groundwater, is notoriously difficult to model.

During the September 24, 2018, LFRG review meeting regarding the suitability of the 2-D STOMP cross-section models used to evaluate releases and vadose transport at a site with significant 3-D features (R1 PA Appendix E, Figures E.1–E.4), the lead modeler for the PA modeling effort stated that the site would be "unmodelable" in 3-D with STOMP. Ultimately, the EMDF was modeled using a 1-D advective and dispersive unsaturated zone transport model (RESRAD-OFFSITE) with assumptions of homogenous waste concentrations and averaged (uniform) release rates across the entire EMDF footprint. R2 PA Section 3.3.2.1 states, "The results of the 2-D implementation are judged to adequately capture the effects of the sloping cell bottoms and sides (berms), variable waste thickness, and modeled variation in vadose zone thickness" but the basis of this judgment is not described and, in fact, the text also notes that computing resource limitations were a factor in relying on a 2-D vadose model.

2.2.7 Evaluation of Peak Dose

The R2 PA (p. ES-4) states: "For long-lived, relatively immobile radionuclides that are significant components of the estimated EMDF inventory (e.g., radionuclides of uranium), PA model saturated zone concentration results beyond 10,000 years also are provided." These results, described in PA Section 4.8, indicate a time of peak uranium groundwater concentrations of approximately 60,000 years using the "base case" RESRAD-OFFSITE model. Neptune's supplemental RESRAD modeling (Appendix B) using PA base case transport assumptions indicate that, at times beyond 15,000 years, uranium isotopes and their progeny are likely to be the primary contributors to radiological dose (Appendix B, Table B1), and that these doses could exceed doses modeled within the 1000-year compliance period (maximum of approximately 2.5 mrem/yr at 600 years) by over two orders of magnitude by model year 35,000 (approximately 650 mrem/yr).

It is also important to note that simply invoking the EMDF PA "base case" erosion rate (PA Appendix C, Table C.6) results in complete removal of the 3.35-meter cover in approximately 23,000 years. Although not shown in Neptune's Review Appendix B, Table B1, modeled doses continued to increase beyond 35,000 years. Base case erosion does not account for undercut erosion, which would likely expose waste sooner. The PA should acknowledge the fact that, although the magnitude and timing of very long-term doses are highly uncertain, the eventual degradation of the EMDF engineered disposal system at some point in the future due to natural processes is a certainty unless the facility is maintained in perpetuity. Hence, the EMDF represents a perpetual environmental liability.

In the R2 PA, the issue of high, long-term doses is readily apparent, as shown in the Probabilistic Total Dose Summary figure presented in the discussion of Critical Issue #1. These doses increase dramatically before 10,000 years.

2.2.8 Surface Drainage D-10W

The influence of D-10W on long-term EMDF performance has not been addressed. The PA proposes rerouting the surface expression of the drainage known as D-10W, which lies west of NT-10 and beneath the proposed EMDF footprint. This is a deeply incised drainage, and yet, like similar drainages at the ORR and indeed within BCV, most of its baseflow is likely beneath the ground surface within the saprolite (ORNL 1997c). The subsurface drainage function of D-10W will potentially remain after construction of the EMDF. In fact, engineered drainage features may be required in the vicinity of groundwater discharge zones in wetland areas along D-10W to lower the water table under EMDF, just as engineered drainage in the vicinity of the NT-4 channel was required to lower the water table beneath the Environmental Management Waste Management Facility (EMWMF). In such case, contaminants reaching the water table could migrate directly to streams with little delay or attenuation.

The influence of D-10W on groundwater flow is apparent in the particle tracking documented in PA Appendix D, Section D.3.4, Figure D.14. This figure shows particle track paths before EMDF construction, and the influence of D-10W is quite strong as a collector of subsurface flows.

Although OREM has proposed filling in the surface expression of D-10W, this does not preclude its subsurface behavior as a drain. The post-construction groundwater modeling would remove the surface expression, but the subsurface influence of D-10W could remain as a deeper constant head boundary. In effect, D-10W would then act as a natural underdrain beneath the EMDF, providing faster pathways to Bear Creek than are present in the proposed model.

A greater concern is that since these regularly spaced drainages have developed naturally in their current locations over many millennia (Tauxe 1998), the same processes that made them would remake D-10W, carving a new deep drainage through the EMDF. This would form from the bottom up, following persistent weaknesses in the underlying bedrock. Simply filling in a deep drainage feature such as D-10W might not preclude it from reforming in the future.

2.2.9 Radionuclide Screening

The following concerns exist regarding screening of radionuclides for the EMDF PA modeling:

- Per the Executive Summary, “Any parent isotope [radionuclide] in the EMDF inventory with a half-life of less than 5 years was screened out from further analysis.” This approach overlooks a potentially important step: Some short-lived parents have long-lived progeny (e.g., Cf252 > Cm248) and the mass of these parents was lost in the analysis. In accounting for progeny, some new radionuclides may also need to be added to the list (e.g., Pm147 > Sm147). This same problem is identified in the explanation of the screening process in PA Appendix B, Section B.3.3.

- A second screening step further eliminates radionuclides "based on a peak dose criterion of 0.4 mrem [in a] year" assuming "exposure via groundwater ingestion only." Note that having excluded 34 radionuclides by this method, a summed dose could be as high as 13.6 mrem in a year. This is an inadequate screening, since exposure pathways in addition to water ingestion can cause potentially significant dose.

- PA Appendix B (Section 3.3) states "Activity concentrations are adjusted for radiological decay to the assumed year of EMDF closure (2047) based on radioisotope half-life and the year of data collection." This results in a potential underestimation of the actual inventory expected in 2047 as presented in Table B.6. It is apparent that ingrowth of short-lived progeny, such as Ra-228 and Th-228, was not considered in developing the radionuclide source term.

2.2.10 Upwards Migration Pathways Other Than Radon Flux

Upwards migration pathways other than radon flux were not considered. Section 3.2.2.2 in the R1 PA states that "The conceptual model of radon flux is limited to vapor-phase diffusion of radon from the waste through the overlying cover materials and release at the cover surface." Although no longer stated explicitly, Appendix H of the R2 PA shows that this assumption has not changed.

This approach neglects the migration of radon parents (e.g., Ra-226), which can be transported by diffusion in a continuous water phase in cover materials. While a highly saturated layer of clay, for example, slows the air phase diffusion of radon gas, it also promotes the water phase diffusion of radium and other parents of radon. Furthermore, the buildup of progeny (Pb-210 and Po-210) resulting from the decay of radon near the ground surface needs to be accounted for in exposure pathways that are dependent on surface soil concentrations, such as external "shine," and contributions through foodstuffs grown or raised on these soils. Upwards diffusion of radionuclides in pore water, and buildup of Pb-210 and Po-210 in the cover, should be included in the PA model.

2.2.11 Waste Characterization

Waste characterization is not adequately addressed:

- The design and modeling of a radioactive and mixed-waste landfill should be informed by the wastes that it is expected to contain. Waste characterization in the PA is based in part on the waste characterization for the Environmental Management Waste Management Facility (EMWMF), though the wastes are fundamentally different (from Y-12 and X-10 rather than K-25) and the waste profiles for EMWMF are themselves incomplete. For example, Cl-36 (which was a significant issue for the Solid Waste Storage Area 6 PA and the Interim Waste Management Facility) has appeared in EMWMF wastewater, but was not analyzed in its PA. Chlorine-36, like I-129, is fast-moving and long-lived.

- It is not clear how the PA intends to treat Cl-36, one isotope for which the PA provides no estimate of the inventory in the waste. Chlorine-36 has been reported as present in

EMWMF wastewater, but neither the intruder scenario, the all-pathways scenario, nor the groundwater base case was performed for Cl-36. The R2 PA states on page 115, “Cl-36 was included only in the Phase 2 screening model using a unit source concentration of 1 pCi/g to provide information for future waste management decisions.” No single radionuclide soil guidelines or other risk-based limits were generated for Cl-36, and there is no explanation of how the screening model results will be used for future waste management decisions.

- The use of EMWMF profiles for ORNL and Y-12 waste is of limited utility because detailed characterization for EMWMF Waste Acceptance Criteria (WAC) compliance was carried out for a restricted number of isotopes. Detailed characterization data for EMWMF waste lots was typically limited to that for uranium isotopes and Tc-99. The PA does not identify which EMWMF waste lots were used in deriving the EMDF waste inventory estimates.

- Some information was to be extracted from the SORTIE database, but the work seems to have created some errors. For example, outliers were excluded based on “lack of information,” but that alone does not justify exclusion. Was it applied to all SORTIE data in an unbiased fashion, or just to these outliers? The process for excluding data should be discussed in more detail. The R2 PA states that “There is considerable uncertainty in the estimated activity inventories of C-14, Tc-99, and I-129, which are the three more mobile dose drivers for the performance analysis.” Inventory estimate uncertainty can cover multiple orders of magnitude. Some isotopes that are likely to be present at some level in the waste are excluded from the inventory due to lack of information. It is not clear whether efforts were made to collect additional characterization data when these data gaps were identified.

- Section 1.7.3 of the R2 PA claims that the estimated radionuclide inventory has a pessimistic bias. Depending on the specific radionuclide in question and on the waste streams ultimately selected for on-site disposal, there is the potential on the ORR for the inventory to be either higher or lower than the inventory reported in the PA. A thorough review of the waste profiles and other data sources referenced in the PA would be necessary to better estimate the uncertainty in the inventory. Technetium-99 is ubiquitous on the ORR and C-14 and Cl-36 have been reported in EMWMF wastewater at levels over 200 pCi/L and 75 pCi/L, respectively. Consequently, the assumption that the estimated post-closure inventory assumed for the purposes of this performance assessment has resulted in a pessimistic bias is not supported.

- Uncertainty in the inventory of disposed radionuclides is likely to be one of the more significant sources of overall uncertainty in the PA results, but this was not considered in the probabilistic UA. The R2 PA Executive Summary states, “the use of the SORTIE [inventory] data should lead to overestimation of average waste concentrations because the facility inventories developed for safety analysis tend to be bounding (maximum likely) estimates.” This statement is both vague and unsupported. In some cases, it can be incorrect, as when progeny not yet present produce higher risks than the parents. Safety analyses are performed for a very different purpose than are PAs, and should not be relied upon for PA inventory development. The degree of such overestimation can reasonably be expected to vary depending on waste stream and radionuclide, and underestimation is also a possibility. A multivariate UA should capture the state-of-knowledge for radiological inventory of the wastes destined for the EMDF in an unbiased, defensible, and reviewable manner.

- The R2 PA acknowledges:

  > There is considerable uncertainty in the estimated activity inventories of C-14, Tc-99, and I-129, which are the three more mobile dose drivers for the performance analysis.

  but asserts:

  > The conclusion is that although post-operational inventory uncertainties for C-14, Tc-99, and I-129 are high, only the assumed EMDF average I-129 activity concentration value applied in the PA models constitutes a key parameter assumption that supports determination of EMDF compliance with the all-pathways performance objective.

  However, the assumed concentrations of the three radionuclides discussed above in as-generated waste are so low (less than 10 pCi/g) that they would be difficult to accurately measure with standard waste characterization methodologies and analytical techniques. Given the widespread distribution of Tc-99 on the Oak Ridge Reservation, characterization efforts sufficient to quantify a post-closure inventory of 1.56 pCi/g might be prohibitively expensive. Given that the PA overestimates travel times to points of assessment, uncertainties of only an order of magnitude in Tc-99 and C-14 post-closure inventories, which seem possible if not probable, could compromise compliance with performance objectives.

- A comparison of inventories of key radionuclides in the R2 PA Section 2 tables shows that inventories of U-235 decreased about threefold and inventories of U-238 decreased about twofold compared with the R1 PA, while U-234 stayed exactly the same (instead, one would expect the ratios to be more or less the same). This indicates that the uncertainty in inventory is not addressed in any meaningful way. Section 1.7.3 of the R2 PA discusses “pessimistic biases,” including this statement for inventory:

  > “Modeled radionuclide inventories are based on the full EMDF waste volume capacity (2.2 million cy), and average activity concentrations for EMDF waste streams are likely over-estimated. The EMDF design capacity incorporates an added 25 percent to the projected CERCLA waste volume (DOE 2017b, Appendix A) to account for volume uncertainty. The approach to estimating activity concentrations in waste is intended to overestimate concentrations to account for uncertainty in the characteristics of future remediation waste (Appendix B). As a result, the activity inventories used in the PA models are higher than inventories likely to be present at EMDF closure.”

  Instead of deliberately biasing estimates, the analysts should strive to make accurate estimates, with appropriate amounts of uncertainty that reflect the state of knowledge.

- Because the PA results are conditioned on the "pessimistic bias" of the key radionuclide inventories that will be disposed, it would be appropriate to develop waste acceptance criteria and waste sampling protocols that ensure that the average waste concentrations of these radionuclides in the EMDF do not exceed the inventories used in the PA modeling. It's important to realize that the R2 PA conclusion that only I-129 inventory uncertainty is important for determination of EMDF compliance with the all-pathways performance objective (Section 1.7.1) is conditioned on assuming significant loss of the as-disposed inventories of C-14 and Tc-99 during landfill operation. As discussed in relation to Critical Issue 5, this assumption is not adequately supported.

- Section 5.1 of the PA states, "Presentation of STOMP model sensitivity evaluations is limited to Tc-99 results, which are representative of the sensitivity of predicted concentrations of other radionuclides with nonzero K_d values (e.g., I-129) to the uncertainties in K_d values." Although cover infiltration applies generally to all radionuclides, no basis is provided for stating that Tc-99 is representative of other radionuclides with respect to uncertainty in K_d. There should be great variability in the degree of confidence in the K_d values among the different chemical elements being modeled.

2.2.12 Land Use and Exposure Scenarios

The land use and exposure scenarios are not adequately defended.

- The text of PA Appendix G, Section G.2.1, states, "The well water is used as a drinking water source and for household activities such as showering. Impacted groundwater also is assumed to pass downgradient of the well and enter a surface water body (Bear Creek) used by the resident farmer to irrigate crops, provide water for livestock, and for recreational fishing." It isn't necessarily protective to evaluate a true "farmer" who uses surface water for irrigating large fields; commercial crop farming on such a scale has little in common with food grown for home consumption. A home garden and fruit trees irrigated with well water, and home-raised poultry, is a more credible and protective scenario for evaluating food pathways when the endpoint is individual dose for a hypothetical resident. Agricultural exposures are more suited to a region-specific population dose assessment rather than the hypothetical individual doses that are regulated under DOE O 435.1.

- An attempt was made to dismiss the upward contaminant transport pathways that contribute to doses from gases and particulates emanating from the cover surface. "Releases to the atmosphere are not calculated in the base case model because of the selected source release model; however, dose from the inhalation of vapors and contaminated dust particles released from the EMDF through the vapor and biointrusion pathways are assessed in separate evaluations (Sect. G.4.4.2)." The vapor and biointrusion pathway analysis, however, is not coupled to important upward transport pathway mechanisms of upward diffusion of radon in pore air and of most other radionuclides in water. These upward pathways tend to contaminate the cover, enhancing the source for the analysis. Further, the biointrusion maximum depth of 1 m selected for the RESRAD-OFFSITE run does not have any information to support it. One would expect a review of native plant root depths and animal burrowing depths to inform this value, but none is presented. Note that enrichment of the cover with radionuclides also contributes to external doses for a receptor on the cover.

Most significantly, assuming a residence is located off-site for the entire performance period is inconsistent with the statement in R2 PA Appendix G, Section G.4.1, that institutional controls are assumed effective for limiting site access for only 100 years post-closure. Although DOE M 435.1 differentiates future on-site and off-site exposures based on present-day administrative boundaries, the IHI analysis required by DOE does include an on-site resident, albeit for a limited period of time exceeding a year. So, doses to an on-site resident in terms of annual exposure (with a maximum allowable dose of 100 mrem in a year) apply and should be evaluated.

- The text of PA Appendix G (Section 3.4.2) states, "Irrigation water use for the various crop fields was simulated at a rate of 0.15 m/year, with 100 percent of the water coming from contaminated portions of Bear Creek. An irrigation rate of 0.015 m/year was specified for the offsite dwelling." Because water pathways are the sole avenues of exposure, a strong basis is needed for assumed irrigation rates. No basis is provided for the crop irrigation rate of 0.15 m/yr, or for the landscaping irrigation rate of 0.015 m/yr.

2.2.13 ALARA analysis

As stated in the Executive Summary and Section 1.5.4 of the R2 PA, an ALARA analysis (to keep doses As Low As Reasonably Achievable) is required for a PA under DOE M 435.1 and under DOE O 458.1. The PA does not provide an ALARA evaluation of collective dose. This is rationalized by stating that, "given the likelihood that BCV and the CBCV site will remain under DOE control indefinitely, there are a limited range of collective exposure scenarios that are credible." This statement is inconsistent with the claim that the site will remain under institutional control only "for the foreseeable future" (Executive Summary) and that institutional controls are assumed effective only for limiting site access for 100 years post-closure (Appendix G, Section G.4.1). It is further stated in Section 1.5.4 that "the disposal options considered and conclusions presented in the EMDF RI/FS and draft Proposed Plan are considered to meet the ALARA requirement for the EMDF PA." The PA should specifically describe how the information presented in the EMDF Remedial Investigation/Feasibility Study (RI/FS) and draft Proposed Plan satisfies the requirements of an ALARA analysis.

This critique is pertinent because Base Case dose to a maximally exposed individual (MEI) is calculated to be 1 mrem/yr, and a 1 mrem/yr MEI dose is a threshold for performing semi-quantitative ALARA analysis. Note also that Fig 5.15 in the R2 PA shows 95th percentile doses of 10 to 20 mrem/yr between 1000 and 2000 yr.

2.2.14 Development of Waste Acceptance Criteria (WAC)

The PA should address development of waste acceptance criteria (WAC) to support benefit-cost evaluation for construction of the facility. The PA (page ES-33) states that "…the FFA parties will approve operating limits, including WAC, and will issue a WAC compliance document prior to EMDF operations." The WAC for the EMDF should be developed from the results of the PA and CA to determine the amounts of radioactive materials that can be disposed at the EMDF while maintaining compliance with the annual dose threshold.

When the expected inventory at a site is well-known, and the relative mixtures of radionuclides that contribute to the sum-of-fractions is established, an initial WAC is readily developed. Potential inventories for the EMDF, however, are quite uncertain. In such a case, an initial WAC can be developed, but it is likely to change during operation. The Area 5 Radioactive Waste Management Site at the NNSS takes this approach, since it is often faced with acceptance decisions for unusual or unique wastes. This “living WAC” is flexible, and depends on what the facility has already accepted. In essence, it evaluates the radiological capacity remaining at the site following ongoing disposals. As each major disposal campaign is completed, the PA is rerun in order to determine the remaining capacity, and estimates of radionuclide contents of potential waste streams are likewise updated. In this way, the site can be used to its optimal capacity while being protective of human health and the environment.

Such an approach could be implemented for the EMDF, and this would probably be beneficial considering this, like the NNSS low-level waste site, would be an active disposal facility. The NNSS runs its PA with a new waste stream, and has results available within 24 hours, partly because its PA fully couples all processes in a systems-level model built using GoldSim. Having such a versatile model available for ongoing decisions regarding waste disposal more than compensates for the effort needed to develop such a model (which is really no more effort than has been undertaken in the EMDF for its system of uncoupled models).

Initial WAC development should be done prior to initiating construction of the facility, rather than prior to beginning waste placement, to support benefit-cost comparison of the EMDF. This prudent methodology is followed in other PAs (DOE 2018; Occhiogrosso et al. 2017; ORNL 1997a, 1997b, 1997c; Shott et al. 2000a; Shott et al. 2000b). Until it has been determined that construction and on-site disposal at the CBCV site is cost-effective, based on WAC limits, construction of the EMDF is performed at risk, potentially resulting in the waste of taxpayer money.

2.3 Critical Issues—Composite Analysis

Many of the critical issues identified in relation to contaminant release, transport, and exposure models for the EMDF PA are applicable to the CA because the CA uses the results of the EMDF PA. Moreover, the same RESRAD-OFFSITE model used for the PA is applied to calculate dose based on surface water concentrations at the confluence of NT-11 and Bear Creek. The following critical issues, specific to the CA, are identified, in addition to those already described as critical issues for the PA.

2.3.1 CA Critical Issue 1: MCL Analysis

According to TDEC, groundwater and surface water protection standards are currently exceeded in Bear Creek Valley (BCV). These exceedances would normally preclude construction of the EMDF, which can only increase levels of groundwater and surface water contamination.

Since surface water and groundwater are intimately intertwined in BCV, all waters need to be shown not to exceed MCL standards, now and in the future, per DOE O 435.1 and 458.1. Although an MCL analysis was performed (Section 4.7), it was limited to the compliance period. DOE O 458.1 does not recognize a compliance period. Furthermore, the MCL analysis should be revisited once critical issues and key findings with the PA are addressed; particularly those that indicate that the PA may be understating travel time and contaminant concentration in groundwater.

2.3.2 CA Critical Issue 2: Lack of Dose Estimate

The CA precludes the required estimation of dose from other contributing sources by substituting CERCLA risk goals. As stated in the CA Executive Summary, the results of previous risk assessments and commitments made in previous Records of Decision (RODs) that included the EMWMF "were used in conjunction with CERCLA risk goals of an ELCR (excess lifetime cancer risk) of 1×10^{-5} during the first 1000 years after facility closure and an ELCR of 1×10^{-4} thereafter to determine the concentrations of contaminants that could be accepted at the facility." In other words, the authors of the EMDF CA presume that these goals will be met due to a written commitment (the ROD). No consideration is given to scenarios where these goals cannot be met.

Even so, such risk goals are not suitable proxies for a comprehensive determination of future doses, as required by DOE Order and Manual 435.1. Stating that releases will be monitored and controlled in a manner that any such goals are achieved assumes not only that institutional control (IC) is maintained throughout the modeling period, but that current legal requirements and funding structures are upheld in perpetuity. These assumptions are in fact explicitly stated in Section 1.5 of the CA: "Perpetual institutional controls and site maintenance were included in the selected remedial action alternative in the Phase I BCV ROD (DOE 2000a) and the EMWMF ROD (DOE 1999c)."

2.3.3 CA Critical Issue 3: Inventory of Other BCV Sources

No methodology or plan is cited in the CA regarding how the risk goals described in the RODs, and used as the basis for the CA, will be achieved. Risk is a function of both the environmental concentrations of specific radionuclides in various media, and assumptions regarding exposure to these media. During the CA presentation to the LFRG Review Team, the presenter acknowledged that the inventory of other BCV sources is poorly understood. Without an understanding of the nature of these sources, a CA cannot be adequately developed.

2.3.4 CA Critical Issue 4: EMWMF Modeling Assumptions

The CA computes the contribution of the EMWMF based on past analyses of facility performance. The updated groundwater and contaminant transport model described in Appendix A of the CA suffers from the same oversimplifications that are discussed in comments on PA modeling. Parameters used in groundwater modeling of upper Bear Creek Valley (Table A.1) and inputs to the PATHRAE code given in Table B.3 are not entirely consistent with those used in modeling the performance of EMDF, but result in similarly unrealistic travel times through groundwater.

Assumptions used to evaluate facility performance in the EMWMF RI/FS, the RI/FS addendum, and more recent assessments made for the purposes of PA maintenance and ROD modifications bias the analysis toward underprediction of near-term impacts on water resources. The performance of the EMWMF cover was assumed to be even more robust than that assumed for the EMDF cover, with infiltration into the facility remaining below one centimeter per year throughout the entire 1000-year compliance period. The possibility of groundwater intrusion into the facility buffer and liner, a matter of some concern for the entire operational history of the facility, was not considered in modeling. Modeling did not consider bathtubbing or the likelihood of a liner breach, which might be higher at EMWMF than at disposal facilities with a more typical design, as the composite liner was penetrated by leachate and leak detection pipes.

The most direct path for contaminants to migrate from the EMWMF to the CA point of assessment (POA) is through the underdrain either directly to Bear Creek or to the Maynardville Limestone, where subsurface transport to Bear Creek will be almost immediate, even for strongly adsorbed radionuclides. The travel time for a contaminant released from the liner system under much of the EMWMF to the POA would be approximately estimated by the time required for travel through unsaturated material to the underdrain, which is much less than the travel time calculated in the EMWMF solute transport models. Another appropriate POA would be below the confluence of Bear Creek with East Fork Poplar Creek, where the public currently fish, and a fish ingestion pathway might involve a number of members of the public rather than just the hypothetical maximally exposed individual.

Past EMWMF analyses have used a lower partition coefficient for uranium isotopes (20 ml/g) than that assumed in the EMDF PA (50 ml/g). However, operational experience at EMWMF suggests that, at the facility scale, even this lower value may result in an underprediction of migration rates in groundwater, not only in the saturated zone but also in the vadose zone. At least some of the chemical forms of uranium in EMWMF waste, presumably those oxidized to the hexavalent state such as UO_2F_2 or uranyl carbonate complexes, were readily leached from waste. Uranium also migrated almost immediately into the leachate collection system through the one-foot protective layer, which was designed to have a hydraulic conductivity of 10^{-5} cm/s or less and was constructed primarily from the same clay rich saprolite that comprises the buffer and vadose zone under the facility. The delay time of weeks or months through the EMWMF protective layer that might have been anticipated from a model of vertical transport with retardation was not observed. Based on this experience, the most significant delay in uranium migration from the EMWMF to the POA might result from that due to migration through the liner system or from the time to failure of landfill components. The travel time for EMWMF-sourced uranium to the POA is likely to be much reduced from that calculated for the dose analysis in the CA.

2.4 Key Findings—Composite Analysis

- The CA includes an incomplete technical basis for identifying the POA for the CA. The CA should present an analysis of other possible POAs, such as locations in downstream reaches of East Fork Poplar Creek and Poplar Creek receiving contributions from BCV, Y-12, and the East Tennessee Technology Park, to demonstrate that the selected POA is appropriate. The purpose of the CA is to consider all other DOE sources of radioactive contamination that would interact with releases from the EMDF. This includes not only other sites within BCV, but across the ORR. Most of the radioactive effluents from the ORR collect in the Clinch River, and full mixing into this large surface water body does not occur until well downstream of the mouth of Poplar Creek. At least one appropriate POA, then, is at the mouth of Poplar Creek where upstream contamination from White Oak Creek, Bear Creek, East Fork Poplar Creek, and Poplar Creek itself are integrated. A comprehensive CA, therefore, would include all sources on the ORR. Such a CA could serve any proposed disposals on the ORR in the future, as well.

- The CA does not describe how, or if, environmental contamination remaining in the subsurface from remediation of the S-3 Ponds, Boneyard/Burnyard, and similar facilities has been accounted for.

- Confidence that the risk targets codified in the RODs will be achieved is undermined by the fact that the CA does not describe any efforts made since the signing of the RODs to remediate these sources of contamination. Page ES-2 states "Measurements of contaminant levels in the groundwater, Bear Creek, nearby springs and seeps, and tributaries to Bear Creek from 2001 to 2017 indicate that some of these sites are releasing radioactive contaminants, principally uranium."

- It may not be practicable to achieve the goals of the Phase I ROD. The goals of the ROD are currently not met, and final remediation of the two primary contributing source areas to the radionuclide flux in Bear Creek has been deferred indefinitely. The flux of uranium currently entering Bear Creek from the Bear Creek Burial Grounds (BCBG) must be significantly reduced to meet ROD goals. However, a large inventory of uranium, estimated to be 18.6 million kilograms in the R2 CA, remains in BCBG.

  The Oak Ridge Site Specific Advisory Board (SSAB Recommendation 195, 2011) has stated: "Because of budget limitations and unknown technical risks and challenges, it is not feasible at this time to contemplate any final remediation of the BCBG." The SSAB requested an evaluation of "more modest, actionable remediation ideas" for BCBG, as have regulatory authorities, but no actions have been taken to curtail the transport of uranium from this source to Bear Creek since the 1990s. The approach taken in the CA to evaluate the potential for future exposure due to DOE sources of contamination in Bear Creek Valley does not consider the uncertainty in DOE's ability to constrain releases from sources other than EMDF and EMWMF, particularly releases from BCBG.

- Section 2.3.9.5 of the CA includes a description of the collection systems used to capture contaminated groundwater seeps at Bear Creek Burial Grounds (BCBG). However, there is no quantitative information on the uranium flux that might show trends in releases from the BCBG, nor is that captured uranium flux added to that in Bear Creek when the current baseline for uranium migration from other sources is evaluated. It should also be noted that, although groundwater paths from the disposal trenches to the seeps are much less than 100 meters, uranium migrated readily from the trenches to nearby tributaries. In contrast, the inputs to PATHRAE-RAD given in Appendix B of the R2 CA for groundwater velocity (4.2 meters per year), bedrock density (1800 kilograms per cubic meter), and porosity (0.05), and the assumption of a partition coefficient for uranium of 50 milliliters per gram as assumed in the EMDF PA, predicts that advective transport of uranium over a distance of just 1 meter would require over 4 centuries.

- Section 2.4.4 of the CA states that "The BCBG (Bear Creek Burial Grounds) were not included in the Phase I BCV ROD and activities to reduce contaminant migration from these sources were deferred to a future decision under CERCLA. A ROD defining these activities has not been prepared." This implies that the BCBG are not subject to the Phase I ROD, or the associated 10^{-5} ELCR risk goal. The CA text continues, "The primary sources of uranium in the shallow groundwater/Bear Creek system and the Maynardville Limestone appear to be BCBG and the secondary sources underlying the S-3 Site (i.e., groundwater plumes)." If the primary sources of uranium in the BCV are not subject to the codified 10^{-5} ELCR risk goal, then this goal cannot be used to represent the future risks or doses that these sources present. The CA should clarify the relationship of the BCBG to the Phase I BCV ROD.

- The CA does not present a technical justification for the assumption that radionuclides will account for 100% of the 10^{-5} ELCR risk value that was used to represent the contribution of upstream sources. Depending on the inventory of these sources, it is conceivable that some or all of the 10^{-5} ELCR risk target could be accounted for by non-radiological constituents at some point(s) in time during the compliance period. This could significantly affect the plausibility of actually achieving this goal.

- The use of PATHRAE, a computer program developed in 1986 by Rogers and Associates Engineering, is not an appropriate choice in this application. This program has not been maintained and is a simple screening-level model unsuited for evaluating release and transport from radiological sites in the complex terrain of BCV. Furthermore, it is not clear that this program meets the software quality assurance requirements of the American Society of Mechanical Engineers (ASME) NQA-1 (nuclear quality assurance) certification program "Quality Assurance Requirements for Nuclear Facility Application".

- The method used in the R2 PA to calculate concentrations in surface water is based on the calculation of a flux through groundwater that is subsequently diluted in the creek. Section 4.2 of the R2 CA states that, "The use of an average flow rate in the creek is considered appropriate because it is assumed that the hypothetical receptor uses water from the creek every day of the year." However, the concentration in the creek—not the flow—would be consumed on a daily basis, so the risk to a hypothetical receptor is inversely proportional to stream discharge. The inverse function is a convex function, and so Jensen's inequality states that the inverse of the mean (concentration as calculated in the CA) will be smaller than the mean of the inverses (average of daily concentrations). The use of the average flow to predict concentration for the purposes of risk may thus significantly underpredict the true average concentration. To check the magnitude of this underestimate, the inverse of the average of Bear Creek daily discharge data from 2007 through 2013 was compared with the average of the inverse of daily discharge. Over this seven-year period, the average of daily concentrations would be 3.3 times the concentration that would be estimated using the procedure followed in the CA.

- In Section 1.4, the CA explains that the location of the proposed EMDF in Central Bear Creek Valley conflicts with the future land use in Bear Creek Valley codified in the Phase I

BCV ROD. The CA states that the EMDF ROD will change land use at the proposed site. However, based on public comments, including a petition signed by many Oak Ridge residents, the proposed change in land use may not be well received by the public.

The CA assumes regulatory and community acceptance of this and other similar DOE proposals. To the extent that the CA conclusions rely on regulatory and community acceptance, these conclusions are premature, at least until the EMDF ROD has been approved.

- In Section 2.3.1, the R2 CA describes the physiography of the Oak Ridge area:

  > The ORR area is characterized by long linear northeast-southwest stream valleys between roughly parallel ridges. These define essentially isolated hydrologic systems (watersheds) with little exchange of water from one watershed to another.

  This is reiterated to justify the restriction of sources and points of compliance/assessment to the area within the Bear Creek watershed in Section 3 of the CA. This ignores the possibility that groundwater may be more readily exchanged between adjacent watersheds that lie within the same valley, either due to underflow through karst pathways or as the result of induced gradients due to groundwater pumping. While water balances and exit pathway monitoring do not indicate significant exchange of groundwater between the Bear Creek watershed and either the Upper East Fork Poplar Creek watershed to the east or the Grassy Creek watershed to the west, both underflow through the karstic Maynardville Limestone and groundwater capture across watershed boundaries has been verified at the east end of the Y-12 plant.

- The CA estimates the peak dose from uranium isotopes to occur at 45,000 years and 79,000 years post closure from EMWMF and EMDF, respectively. As stated in earlier comments on key assumptions, the modifications to EMWMF design, the physical and chemical form of a considerable part of the disposed uranium inventory, and travel times for peak arrival measured in a tracer test conducted by TDEC after approval of the EMWMF ROD would indicate that the uranium flux from EMWMF will impact Bear Creek soon after failure of engineered landfill components. The time for significant failure of landfill hydraulic barriers is uncertain and might reasonably be thought to occur at any time within a few decades to many centuries, but it is unreasonable to suggest that clay and plastic will hydraulically isolate waste for millennia. Much of the delay until peak dose results from unrealistic modeling assumptions that fail to properly define the most probable paths for contaminants to reach surface water and do not realistically represent groundwater flow and solute transport in Bear Creek Valley. The conceptual site model in the CA provides an adequate summary of site conditions, acknowledging rapid flow of groundwater in fractures and solution conduits; however, this is not reflected in the equations and input parameters that form the basis for the transport modeling used in PATHRAE-RAD.

  Likewise, uranium at the EMDF is assumed to migrate 300 meters through groundwater to reach surface water, which is not likely to be the most probable or the most rapid path for contaminants to reach the POA for the CA. A more realistic flow path to the CA POA from the EMDF would be migration from the landfill parallel to the valley axis (and geologic strike) to NT-11, then flow down NT-11 to the confluence with Bear Creek (the POA). As noted above, experience gained from many decades of studies of solute transport in the Maryville and Nolichucky formations is not reflected in either the equations describing solute transport, or in the input parameters used in the RESRAD-OFFSITE modeling exercise. If transport models based on simple analytical solutions to the advection dispersion equation such as those in PATHRAE-RAD or RESRAD-OFFSITE are to be used for estimating solute transport, they should be calibrated with tracer results and supplemented with discrete fracture modeling. Otherwise, the results will potentially produce absurdly long travel times, such as those reported in this CA.

2.5 Additional Concerns Beyond DOE O 435.1

In the course of reviewing the PA and CA against DOE O 435.1, a concern that is not strictly regulated under this order is also identified.

2.5.1 Additional Critical Concern 1: Risks from Non-Radiological Wastes

Section 1.3 of the R2 PA states:

> The EMDF preliminary design satisfies Resource Conservation and Recovery Act of 1976 (RCRA) and Toxic Substances Control Act of 1976 design requirements for hazardous and toxic waste disposal units.

Section 1.5.5.2 (non-DOE requirements) states, "Land Disposal Restrictions (LDRs) per 40 CFR 268 will be an ARAR for EMDF disposal of waste containing hazardous constituents above regulatory limits (e.g., for mercury)." … "Future EMDF annual summary reports will include external regulatory requirements that are relevant to PA assumptions and/or the modeling approach. As part of the development of annual summary reports for the EMDF, proposed activities, new ARARs, or other new information that could challenge key assumptions for the EMDF performance analysis will be evaluated in accordance with the EMDF change control process to assess the potential for such changes to require a Special Analysis or revisions to the PA."

The regulatory and scientific bases offered in the PA for excluding evaluation of potential risks from non-radiological waste constituents are not convincing. The wastes to be disposed at the EMDF include low-level radioactive waste (LLW), RCRA hazardous waste, Toxic Substances Control Act (TSCA) PCB wastes, and mixtures of these wastes. Application of contaminant transport models to non-radiological waste constituents would support scientific evaluation of RCRA and TSCA objectives for protection of human health. Regardless of the applicability of specific limits for waste disposal, such as leachate concentrations under RCRA treatment, storage, and disposal rules, an understanding of potential long-term non-radiological impacts specific to the EMDF can only enhance informed waste disposal decisions.

In addition to the need for assessing the impacts of non-radiological hazardous constituents, it is also important to recognize that the nephrotoxic effects of uranium are much more sensitive than radiological dose. As shown in a Technical Memorandum (Neptune, March 2019), water concentrations of uranium associated with adverse nephrotoxic effects are approximately 50 times (natural U) to 100 times (depleted U) lower than concentrations associated with a

25 mrem/yr dose. Consequently, applying a threshold radiological dose as the health-based standard for uranium exposure is not protective of human health for either natural or depleted uranium.

2.5.2 Key Findings for Additional Critical Concern 1: Risks from Non-Radiological Wastes

- Mercury is expected to be the most significant contributor to non-radiological constituent risks in the waste stream identified for disposal at the EMDF. Neptune's Supplemental RESRAD transport modeling (Appendix C) used RESRAD-OFFSITE's default mercury K_d value of 52 cm^3/g in the Base Case model and the long-term degraded infiltration rate of 1 in/yr from the PA. Breakthrough to groundwater for mercury occurred at approximately model year 4,600, and surface water concentrations reached 15 ng/L (maximum) at model year 25,000. Supplemental modeling using an assumed infiltration rate of 2 in/yr resulted in mercury surface water concentrations reaching 1 ng/L around year 15,000 and 50 ng/L around year 19,000. These simulations used the RESRAD-OFFSITE Version 2 release, with the leach rate calculated internally based on the K_d.

- In Appendix C, the RESRAD default mercury K_d of 52 cm^3/g was replaced with a value of 11.6 cm^3/g to gauge sensitivity of mercury transport to uncertainty in the mercury K_d. With 1 in/yr of infiltration, surface water mercury concentrations reach 50 ng/L at year 5500, and peak at about 430 ng/L around model year 8500.

- If elemental mercury is present in the waste, the supplemental simulations are likely to be inapplicable, as elemental mercury is insoluble in water but may migrate as a vapor or as a free-phase liquid. The chemical form of mercury released from the waste—particularly whether it is as elemental mercury or as a chemical compound—will therefore greatly influence its fate and transport. Further, the nature of any treatment that may be utilized for disposed mercury-containing waste, the longevity of such treatment, and environmental conditions within the disposal facility can also affect the release and transport of disposed mercury.

- Uranium is known to have toxic effects independent of its radiological properties. EPA has published oral reference dose values pertaining to the nephrotoxic effects of uranium under CERCLA and the Safe Drinking Water Act. The toxic effects of uranium could readily be evaluated within a coupled probabilistic PA model.

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A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee:

Appendix A
Bathtub Analysis Review

1. Title: A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee: Appendix A Bathtub Analysis Review		
2. Filename: EMDF PA Review Appendix A_FINAL.docx		
3. Description:		
	Name	Date
4. Originator	Ralph Perona	21 Nov 2018
5. Reviewer	Terry Jennings, Paul Black, and Chris Schaupp	3 Dec 2018
6. Remarks		

CONTENTS

TABLES

A1.0 Introduction

This review addresses the evaluation of a "bathtub scenario" presented in Section C.3 of Appendix C of the EMDF PA. As noted in the second paragraph of Section C.3, "This type of scenario could occur if the cover system and/or any post-closure leachate management systems were to fail or degrade, but the liner system continues to perform in accordance with the design specification."

Flaws discovered in the Bathtub Analysis were summarized as Critical Issue 2 of the Neptune and Company, Inc. (Neptune) review of the EMDF PA and CA. This appendix provides technical details in support of that summary. As discussed below, these flaws include,

a. failure to demonstrate that the probability of bathtubbing is negligible, and therefore that it is appropriate to address this situation as a stand-alone, separate analysis rather than as a component of the base case,

b. failure to evaluate the ingrowth of short-lived radionuclides in decay series during the modeling period,

c. failure to assess the potential consequences of bathtubbing by comparing calculated surface water concentrations to applicable standards, and,

d. failure to evaluate the cumulative risk of bathtubbing for all inventory radionuclides.

A2.0 Key Findings

The results of Neptune's review of the bathtub analysis in Appendix C, Section C.3 of the EMDF PA is itemized in five points below. The scope of the review includes the conceptual basis for the bathtub analysis as well as the methods used to estimate and interpret future leachate concentrations under bathtub conditions. This review concludes with an example table indicating how the significance of modeled future surface water concentrations can be evaluated using published standards and a sum-of-ratios approach. This review does not encompass evaluation of the surface water mixing model used to derive the surface water mixing (dilution) factors shown graphically in PA Appendix C, Figure C.14.

1. To support evaluation of bathtubbing as a "sidebar" analysis in an appendix, the analysis should demonstrate that the probability that waste saturation could occur due to the degradation of the engineered system is essentially negligible. To that end, information on both the probability and potential consequences of bathtubbing should be provided. As noted in Appendix C, Section C.3 of the EMDF PA, the probability of some degree of bathtubbing is related to a condition where infiltration through the cover exceeds the flow of water through the liner system below the waste. Logically, degradation of the geomembrane and clay liner in the above-ground cover can be anticipated to proceed faster than below-grade features that are unaffected by processes such as waste settling and the impacts of severe meteorological events. Therefore, it is reasonable to conclude that some degree of bathtubbing is likely in the future, and so this phenomenon should be evaluated within the base case model. Lastly, the bathtubbing analysis should include or reference information demonstrating the physical stability of the engineered landfill under conditions where the waste is saturated. Section 2.2.4 of the PA (Structural Stability) notes that analysis of structural stability is outside the scope of "conceptual design," but this is immaterial because the PA analysis is not based on a conceptual design, it in fact applies a specific engineering design.

2. Review of Appendix C, Table C.6, shows that concentrations of each individual radionuclide have been decayed over time but that ingrowth has incorrectly been ignored, creating contaminant mass balance errors. In the case of radionuclides involved in decay chains, particularly the uranium, thorium, actinium and neptunium series, shorter-lived radionuclides achieve secular equilibrium with longer-lived parents. The error in not accounting for both decay *and* ingrowth resulted in incorrect future concentrations of certain radionuclides. RESRAD-ONSITE v7.2 (with the evapotranspiration coefficient set to ~1.0 to simulate a static contaminated zone) was used to calculate radionuclide concentrations over time using the initial concentrations of radionuclides listed in Table C.6. In Table A 1, yellow highlighting is used to identify radionuclides with concentrations that are obviously lower than those of their longer-lived parents. Pink highlighting shows those radionuclides, in addition to those highlighted in yellow, that will also experience ingrowth from decay of parents listed in the table.

Table A 1. Decayed EMDF Inventory Concentrations.

Radionuclide	Initial Conc (Table C.6)	Decayed Conc (250 y)	Decayed Conc (750 y)
	pCi/g	pCi/g	pCi/g
Ac-227	2.17E-03	7.96E-01	2.06E+00
Am-241	5.50E+01	4.15E+01	1.87E+01
Am-243	5.44E+00	5.31E+00	5.07E+00
C-14	2.93E+00	2.84E+00	2.67E+00
Cd-113m	4.28E-01	1.97E-06	4.16E-17
Cf-249	1.07E-06	6.71E-07	2.50E-07
Cf-250	7.40E-06	1.31E-11	4.06E-23
Cf-251	2.10E-07	1.73E-07	1.18E-07
Cf-252	1.05E-05	3.88E-34	0.00E+00
Cm-243	3.81E-01	9.85E-04	6.63E-09
Cm-244	1.11E+02	7.72E-03	3.73E-11
Cm-245	3.33E-02	3.26E-02	3.13E-02
Cm-246	1.59E-01	1.53E-01	1.43E-01
Cm-247	7.80E-03	7.80E-03	7.80E-03
Cm-248	1.11E-02	1.11E-02	1.11E-02
Co-60	2.25E-02	1.19E-16	2.80E-45
Cs-134	7.14E-09	2.80E-45	0.00E+00
Cs-135	4.21E-01	4.21E-01	4.21E-01
Cs-137	1.24E+03	3.97E+00	4.07E-05
Eu-152	2.67E+01	7.36E-05	5.60E-16
Eu-154	5.71E+00	9.97E-09	3.04E-26
Eu-155	5.04E-02	7.87E-18	0.00E+00
Fe-55	4.49E-07	1.43E-34	0.00E+00
H-3	1.04E+01	2.79E-07	8.67E-20
I-129	3.40E-01	3.40E-01	3.40E-01

Radionuclide	Initial Conc (Table C.6)	Decayed Conc (250 y)	Decayed Conc (750 y)
K-40	2.84E+00	2.84E+00	2.84E+00
Na-22	8.17E-07	9.73E-36	0.00E+00
Nb-93m	0.00E+00	4.87E-01	4.87E-01
Nb-94	9.63E-01	9.55E-01	9.38E-01
Ni-59	3.04E+00	3.04E+00	3.02E+00
Ni-63	3.61E+02	6.39E+01	2.01E+00
Np-237	3.30E-01	3.34E-01	3.39E-01
Pa-231	2.39E-01	8.76E-01	2.14E+00
Pb-210	3.19E+00	8.57E-01	1.55E+00
Pd-107	1.41E-02	1.41E-02	1.41E-02
Pm-146	1.64E-04	3.94E-18	2.80E-45
Pm-147	1.25E-02	2.57E-31	0.00E+00
Pu-238	8.38E+01	1.16E+01	2.23E-01
Pu-239	5.41E+01	5.38E+01	5.31E+01
Pu-240	5.77E+01	5.65E+01	5.36E+01
Pu-241	2.04E+02	3.38E-02	3.14E-02
Pu-242	1.72E-01	1.72E-01	1.72E-01
Pu-244	3.68E-03	3.68E-03	3.68E-03
Ra-226	7.12E-01	8.85E-01	1.61E+00
Ra-228	2.17E-02	3.38E+00	3.38E+00
Re-187	1.71E-06	1.71E-06	1.71E-06
Sb-125	2.39E-04	1.53E-31	0.00E+00
Se-79	9.62E-02	9.61E-02	9.59E-02
Sm-151	4.19E+02	6.11E+01	1.30E+00
Sn-126	1.52E-01	1.52E-01	1.52E-01
Sr-90	3.94E+02	9.58E-01	5.67E-06
Tc-99	2.97E+00	2.97E+00	2.96E+00
Th-228	1.21E-06	4.23E+00	3.39E+00
Th-229	5.71E+00	6.55E+00	8.16E+00
Th-230	1.66E+00	3.10E+00	5.96E+00
Th-232	3.38E+00	3.38E+00	3.38E+00
U-232	1.02E+01	8.25E-01	5.39E-03
U-233	4.16E+01	4.16E+01	4.15E+01
U-234	6.28E+02	6.28E+02	6.27E+02
U-235	1.21E+02	1.21E+02	1.21E+02
U-236	9.00E+00	9.00E+00	9.00E+00
U-238	1.82E+02	1.82E+02	1.82E+02
Zr-93	5.00E-01	5.00E-01	5.00E-01

3. Neptune was unable to duplicate the initial leachate concentrations shown in Appendix C, Table C.6 of the EMDF PA using the equation shown in Appendix C, Section C.3.5. The equation used to derive concentrations in leachate (C_L, pCi/L) from initial waste concentrations ($C_{initial}$, pCi/g) requires that concentrations be solved for the solid phase of waste (C_s, pCi/g). The equations, and static values for porosity (Por), waste saturation (Sat), and waste particle density ($Dens_{part}$) obtained from Section C.3.5 are shown in the following image:

$C_{initial}$ (pCi/g) = (C_L * Por * Sat) + (C_s * (1-Por))			
$C_{initial}$ (pCi/g) = ((C_s/(K_d * $Dens_{part}$)) * Por * Sat) + (C_s * (1-Por))			
C_L (pCi/L) = (C_s * g/L)/(K_d * $Dens_{part}$)			
K_d	rad specific	ml/g	Table G.11, base case
$Dens_{part}$	2.65	g/cm^3	Section C.3.5
Por	0.419		Section C.3.5
Sat	1.0		Section C.3.5
g/L	1000	g/L water	unit conversion, water

Initial C_s concentrations of several radionuclides were calculated using Excel's Solver routine using the equation in line 2 of the image shown above, and C_L was then calculated according to the equation shown in line 3.

Table A 2 shows a comparison of the leachate concentrations shown in Table C.6 and those calculated by Neptune for several radionuclides. Leachate concentrations for future times were calculated by Neptune using, as a multiplier, the ratio of concentrations at a future time to initial concentrations from Table A 1.

Table A 2. Comparison of Leachate Concentrations Over Time.

	Table C.6 (pCi/L)			Neptune calculations (pCi/L)		
Radionuclide	C_L at T=0	C_L at T=259	C_L at T=720	C_L at T=0	C_L at T=250	C_L at T=750
Ac-227	1.45E-03	3.97E-07	1.60E-13	9.4E-04	3.44E-01	8.91E-01
Am-241	5.50E+01	3.63E+01	1.73E+01	3.6E+01	2.69E+01	1.21E+01
Ra-226	2.37E-01	2.12E-01	1.74E-01	1.5E-01	1.91E-01	3.47E-01
Ra-228	7.24E-03	2.00E-16	1.46E-40	4.7E-03	7.31E-01	7.31E-01

4. Text on PA Appendix C, page C-42 states that it is estimated it will require 720 years post-closure to achieve waste saturation, "assuming 41.9% total porosity and zero initial relative saturation for the water." The text goes on to note that earlier saturation would be expected "if the residual moisture content in the waste zone is considered." Why was a realistic residual moisture value not used? A range of potential moisture contents for the waste at the time when the HDPE membrane is assumed to fail should be documented and used for this analysis.

5. The last paragraph of Appendix C, Section C.4.5, states that the concentrations in Table C.3.6 "may be compared to applicable water quality criteria such as DOE Derived Concentrations Standards," but no such comparisons were found. Applying approximate surface water mixing (dilution) factors from Figure C.14 of 3,000 (approximately year 250) and 1,700 (approximately year 750), the surface water concentrations shown in Table A 3 were obtained. Table A 3 compares surface water concentrations to Derived Concentrations Standards (DOE-STD-1196-2011, Table 5), and sums the ratios of concentrations to standards as a measure of potentially significant impact. Summed ratios for all radionuclides are required to determine whether surface water impacted by the EMDF in a bathtub scenario presents a potentially unacceptable cumulative risk for all inventory radionuclides.

Note that the comparison to Derived Concentrations Standard is for example purposes; other state and/or federal water quality criteria may also be applied.

Table A 3. Example of a Comparison of Modeled Future Surface Water Concentrations to Standards, and Evaluation of Cumulative Risk.

Radionuclide	C_L at T=250	C_L at T=750	C_{sw} at T=250	C_{sw} at T=750	DCS	Ratio,T=250	Ratio,T=750
Ac-227	3.44E-01	8.91E-01	1.15E-04	5.24E-04	0.10	1.15E-03	5.24E-03
Am-241	2.69E+01	1.21E+01	8.97E-03	7.11E-03	0.17	5.28E-02	4.18E-02
Ra-226	1.91E-01	3.47E-01	6.38E-05	2.04E-04	0.087	7.33E-04	2.35E-03
Ra-228	7.31E-01	7.31E-01	2.44E-04	4.30E-04	0.025	9.75E-03	1.72E-02
sum						0.064	0.067

A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee:

Appendix B
RESRAD-ONSITE Supplemental Evaluation for the EMDF PA Dose Assessment: Onsite Resident

1. Title: A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee: Appendix B RESRAD-ONSITE Supplemental Evaluation for the EMDF PA Dose Assessment: Onsite Resident		
2. Filename: EMDF PA Review Appendix B_FINAL.docx		
3. Description:		
	Name	Date
4. Originator	Ralph Perona and Chris Schaupp	21 Nov 2018
5. Reviewer	Terry Jennings and Paul Black	6 Dec 2018
6. Remarks		

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FIGURES

TABLES

B1.0 Introduction

Neptune performed a supplemental evaluation of radiological dose for a hypothetical residential receptor to evaluate the EMDF PA's Chronic Post-Drilling "inadvertent human intruder" (IHI) scenario assumptions and results detailed in Appendix I of the PA. This evaluation supports several Critical Issues described in Neptune's review of the EMDF PA and CA. Specifically, aspects of this evaluation are referenced in Critical Issues 3, 4, 5, and 6 of the review. This review also supports several Additional Key Findings discussed in the review, including:

- The PA does not provide an evaluation of "peak dose" and
- The land use and exposure scenarios, and many of the associated parameter input values, are not adequately defended and are not necessarily protective.

Critical review comments of the EMDF PA exposure model include the observation that garden and livestock exposures would be expected to be larger were a domestic groundwater well used to irrigate a home garden and small livestock, instead of assuming the use of surface water for larger-scale agriculture. Also, as described below, groundwater exposure pathways are logically associated with a resident IHI scenario that includes exposure to the drill cuttings from a water well. Therefore, Neptune's supplemental dose assessments described here pertain to a hypothetical resident located on the EMDF landfill, who utilizes a domestic groundwater well for home and garden purposes. The RESRAD-ONSITE computer code is used for the supplemental evaluation, and employs the same physical characteristics of the waste zone, vadose zone, and saturated zone as the EMDF PA RESRAD-OFFSITE base case model.

The Chronic Post-Drilling IHI scenario described in Appendix I of the PA Report only evaluates exposure pathways related to mixing of drill cuttings from a water well into garden soil. DOE-STD-5002-2017 is referenced in Section I.3 as the basis for excluding all groundwater transport pathways. However, DOE-STD-5002-2017 does not specifically define the pathways for a Chronic Post-Drilling IHI scenario. Section 2.2.8 of DOE-STD-5002-2017 alludes to a DOE scenario when stating, "The DOE chronic scenario uses a dose measure of 100 mrem/year, but excludes the contributions from drinking contaminated groundwater." Note that this statement does not support exclusion of *all* groundwater pathways, and DOE-STD-5002-2017 does not elsewhere describe the attributes of the DOE chronic IHI scenario alluded to. Critically, Section 2 of DOE-STD-5002-2017 provides guidance for conducting a PA and explicitly states that it does not impose requirements. No logical basis is provided in the EMDF PA for excluding evaluation of groundwater pathways in a chronic post-drilling residential IHI scenario.

B2.0 Scope and Objectives of the Supplemental Resident IHI Evaluation

Several key uncertainties identified during review of the EMDF PA are investigated as part of this supplemental evaluation. The objective of these evaluations is to understand the potential significance of certain bounding assumptions on radiological doses over time. The scope of this supplemental evaluation includes:

1. Evaluation of doses for a complete Chronic Post-Drilling IHI scenario, where this scenario includes all realistic (groundwater) exposure pathways in addition to exposure from drill cuttings spread on the ground surface. This evaluation includes development of realistic exposure parameters for home-produced foods, including garden produce and chicken meat and eggs from domestic poultry.
2. Evaluation of the significance of uncertainty in the “base case” Chronic Post-Drilling IHI scenario assumptions for element-specific solid-water partition coefficient (K_d) values, time of geomembrane failure, and long-term performance (infiltration rate) of the engineered cover. Uncertainty in K_d values was simplistically evaluated by replacing “base case” values with recommended K_d values (geometric mean) for a clay soil type (most analogous to shale) from the 2015 RESRAD *Data Collection Handbook* (DCH). Uncertainty in the long-term infiltration rate was evaluated by using the native recharge rate of 8 inches per year to reflect substantial failure (naturalization) of the engineered cover.

B3.0 Key Findings

- Changing the failure time for the geomembrane does not materially affect the shape of the dose curves, it only delays the time when long-lived and mobile radionuclides (such as C-14, I-129, and Tc-99) are leached into groundwater (see Figure B 1). One perspective on this is that by delaying the time when mobile radionuclides are leached, the use of geomembranes thereby increases the probability that releases will not be detected at the time when these releases occur. In other words, the use of geomembranes could actually increase the potential for future exposures by sufficiently delaying the time when such exposures might occur at a time that monitoring is no longer being performed (i.e., institutional control is lost).
- There is a substantial difference in near-term (<1,000 years) cover performance when substituting RESRAD K_d values for the base case K_d values (see Figure B 1 and Table B 1). Although base case K_d values are lower (less “conservative”) than RESRAD values for most elements, the opposite is true for some relatively soluble elements (H, C, and Tc, as well as U).
- Drill cuttings dose decreases to levels below a few mrem per year after a few 100s of years as relatively short-lived fission products like Sr-90 and Cs-137 decay (see Figure B 4). Drill cuttings exposures are relatively unimportant for the Chronic Post-Drilling IHI scenario in comparison to the groundwater pathways exposures.
- The EMDF PA assumption that the long-term performance of the engineered cover will restrict infiltration to 1 inch/year, in conjunction with the assumption of relatively large K_d values for the long-lived soluble elements (C and Tc) and uranium, was critical to the PA conclusion that water-pathways doses from the EMDF will be well below performance metrics. If infiltration rates exceed the PA base case assumption of 1 inch/year, and if some fraction of the inventory of uranium and long-lived soluble radionuclides has lower K_d values than assumed for the base case, it is not unlikely that doses could exceed the 25 mrem/year performance threshold shortly after 1,000 years (see Figure B 5 and Table B 3).
- The nonlinear relationship of infiltration rate and K_d on leaching is indicated by the different timing and magnitude of water-pathways results from substituting RESRAD K_d

values for the base case values when infiltration is 1 inch/year (see Figure B 1, Figure B 2, and Figure B 3, and Table B 1 and Table B 2) versus 8 inches/year (see Figure B 5 and Table B 3). These results highlight and demonstrate a significant inadequacy in the EMDF PA probabilistic analysis, which is that uncertainties in the K_d values and leach rates were treated independently.

B4.0 Results

B4.1 Resident EMDF Receptor

Beginning with the concentrations listed in PA Appendix G, Table G.9 (EMDF Source Concentration), a simple ingrowth/decay model was run in GoldSim simulation software to estimate EMDF source term concentrations at 100, 200, and 300 years—three hypothetical time points for potential geomembrane failure. Corresponding concentrations were then entered into the RESRAD-ONSITE model to calculate total dose at time points ranging from 100 to 50,000 years post-site closure. In addition, two different K_d scenarios were examined for each time point and membrane failure scheme; the two K_d scenarios take values from the "base case" and DCH lists by element in Appendix G, Table G.11 (Other RESRAD-ONSITE inputs are listed in the appendix of this report.) Figure B 1 displays total dose to a hypothetical receptor at the EMDF for eight scenarios representing four starting inventory concentrations (0, or initial, which used EMDF concentrations listed in Appendix G, Table G.11, and 100, 200, and 300 years, corresponding to EMDF concentration estimates from modeling of nuclide decay/ingrowth at 100, 200, and 300 years, based on the aforementioned initial EMDF concentrations) and two K_d scenarios (base case K_d vlaues and RESRAD DCH K_d values). The results of these RESRAD runs indicate that the time of membrane failure is relatively inconsequential with regards to peak dose and time. Though there is a marked difference in total dose within 1,000 years of site closure between the K_d scenarios, total dose in each instance remains relatively low (Figure B 1).

The very long-term contribution (35,000 years) of water-dependent exposure pathways for a receptor situated on the EMDF landfill is shown in Figure B 2. In the base case K_d scenario, water-dependent pathways dose becomes notable on the y-axis scale around 15,000 years post-site closure, and earlier (about 10,000 years post-site closure) when using the RESRAD DCH K_d values. The lower K_d values for C-14, I-129, and Tc-99 in the DCH K_d scenario result in a peak dose that is larger and arrives sooner than occurs in the base case K_d scenario. These simulations make clear that the choice of partition coefficients is immensely important, and as shown in Figure B 2, the model predicts that when DCH-recommended K_d values are used, total dose peaks within 15,000 years post-closure. Note that by approximately model year 25,000, with the assumed 0.000066 m/yr sheet erosion rate (see Section B5.5), more than half of the original cover thickness will have been removed, and cover performance assumptions related to infiltration (1 inch/year) will therefore likely be invalid. For this reason, the estimated doses at times in the very distant future that are shown in Figure B 2 should not be interpreted as predictive of landfill performance and precise total dose. The purpose of simulating landfill performance in the very distant future is to understand the implications of model assumptions on the system behavior, and dose is a convenient metric for integrating such behavior across multiple radionuclides.

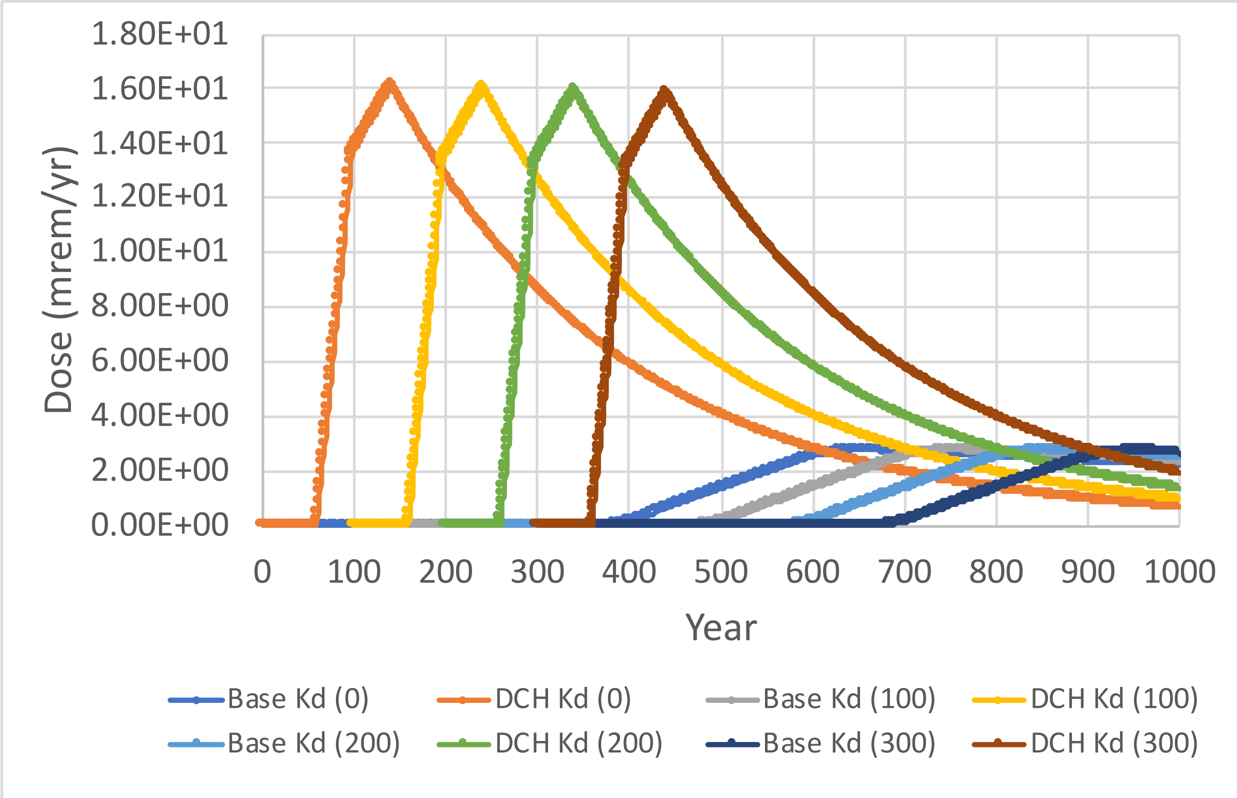

Figure B 1. Integration of the total water-pathways dose estimates to a hypothetical resident from RESRAD-ONSITE model runs for ~1,000 years following the year of potential geomembrane failure (100, 200, 300 yrs.) and for each K_d scenario.

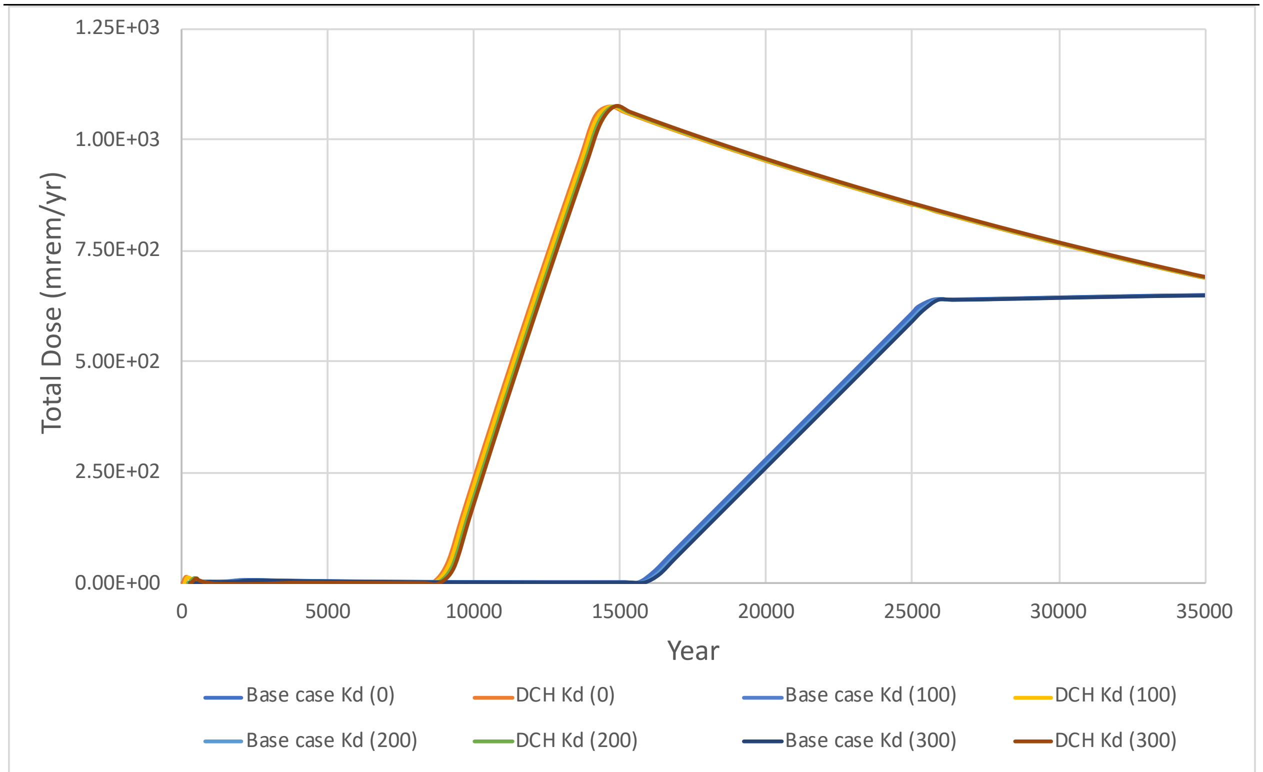

Figure B 2. Integration of the total water-pathways dose estimates to a hypothetical resident from RESRAD-ONSITE model runs for 35,000 years following the year of potential geomembrane failure (100, 200, 300 yrs.) and for each K_d scenario.

Table B 1. Total water-pathways dose and primary contributing radionuclides in the resident EMDF scenario (using initial, or year 0 starting concentrations), for each K_d condition, over 35,000 years. The total dose peak and pattern of nuclides, i.e. when particular radionuclides become significant contributors to total dose, is nearly identical for these K_d scenarios when examined for the cases in which the geomembrane fails at 100, 200, and 300 years; the total dose curve is effectively shifted 100, 200, and 300 years forward in these scenarios.

Base Case K_d			
Year (post-closure)	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*
0	~0	N/A	~0
100	~0	N/A	~0
200	~0	N/A	~0
400	0.16	Tc-99	0.16
500	1.39	Tc-99	0.75
		C-14	0.64
600	2.55	Tc-99	1.31
		C-14	1.25
1,000	2.22	C-14	1.19
		Tc-99	1.03
5,000	3.42	I-129	3.27
10,000	1.40	I-129	1.35
14,294	1.12	I-129	0.64
		K-40	0.35
15,000	1.15	I-129	0.56
		K-40	0.40
20,000	280	U-234	173
		U-238	51.3
		U-235	33.1
25,000	609	U-234	362
		U-238	108
		U-235	69.8
		Pb-210	24.6
30,000	643	U-234	350
		U-238	106
		U-235	68.2
		Pb-210	58.8
35,000	648	U-234	322
		U-238	98.3
		Pb-210	93.8
		U-235	63.4

RESRAD DCH K_d			
Year (post-closure)	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*
0	~0	N/A	~0
100	13.7	C-14	12.3
		Tc-99	1.44
200	12.9	C-14	7.73
		Tc-99	5.16
400	5.93	C-14	3.05
		Tc-99	2.88
600	2.81	Tc-99	1.61
		C-14	1.21
1,000	0.69	Tc-99	0.51
5,000	2.74	I-129	2.74
10,000	241	U-234	153
		U-238	44.5
14,294 (peak)	1.08×10^3	U-234	687
		U-238	201
		U-235	130
		U-233	45.6
15,000	1.07×10^3	U-234	674
		U-238	198
		U-235	127
		U-233	44.6
20,000	952	U-234	586
		U-238	174
		U-235	112
		U-233	38.4
25,000	853	U-234	509
		U-238	152
		U-235	98.2
		Pb-210	39.4
		U-233	33
30,000	765	U-234	443
		U-238	134
		U-235	86.3
		Pb-210	52.3
35,000	688	U-234	385
		U-238	118
		U-235	75.7
		Pb-210	62.8

A comparison of the results of the supplemental evaluation Neptune performed using RESRAD-ONSITE and the EMDF PA "base case" results developed using RESRAD-OFFSITE is displayed in Figure B 2. Though the overall radionuclide profile over 10,000 years is relatively similar between the Neptune and the EMDF PA results, the time at which important nuclides, such as C-14, Tc-99, and I-129, begin to contribute significantly to receptor dose differs greatly (Table B 2). Moreover, peak dose in Neptune's evaluation occurs at approximately year 2,300, more than 3,000 years earlier than in the EMDF PA evaluation (Table B 2). Mobile radionuclides Tc-99 and C-14 are the major contributors to dose for the first ~1,000 years in the RESRAD-ONSITE run (represented by first rise and peak in Figure B 2; at ~600 years they contribute to dose equally, see Table B 2), with I-129 dominating for the next ~9,000 years (represented by the large peak beginning at ~ year 1,200, peaking at year 2,326, and decreasing until year 10,000 in Figure B 3). In contrast, the EMDF PA RESRAD-OFFSITE run indicates that the mobile radionuclides H-3, C-14, and Tc-99 are the largest contributors to dose for the first ~5,000 years, and I-129 does not begin to contribute significantly to dose until ~year 5,000 (Table B 2).

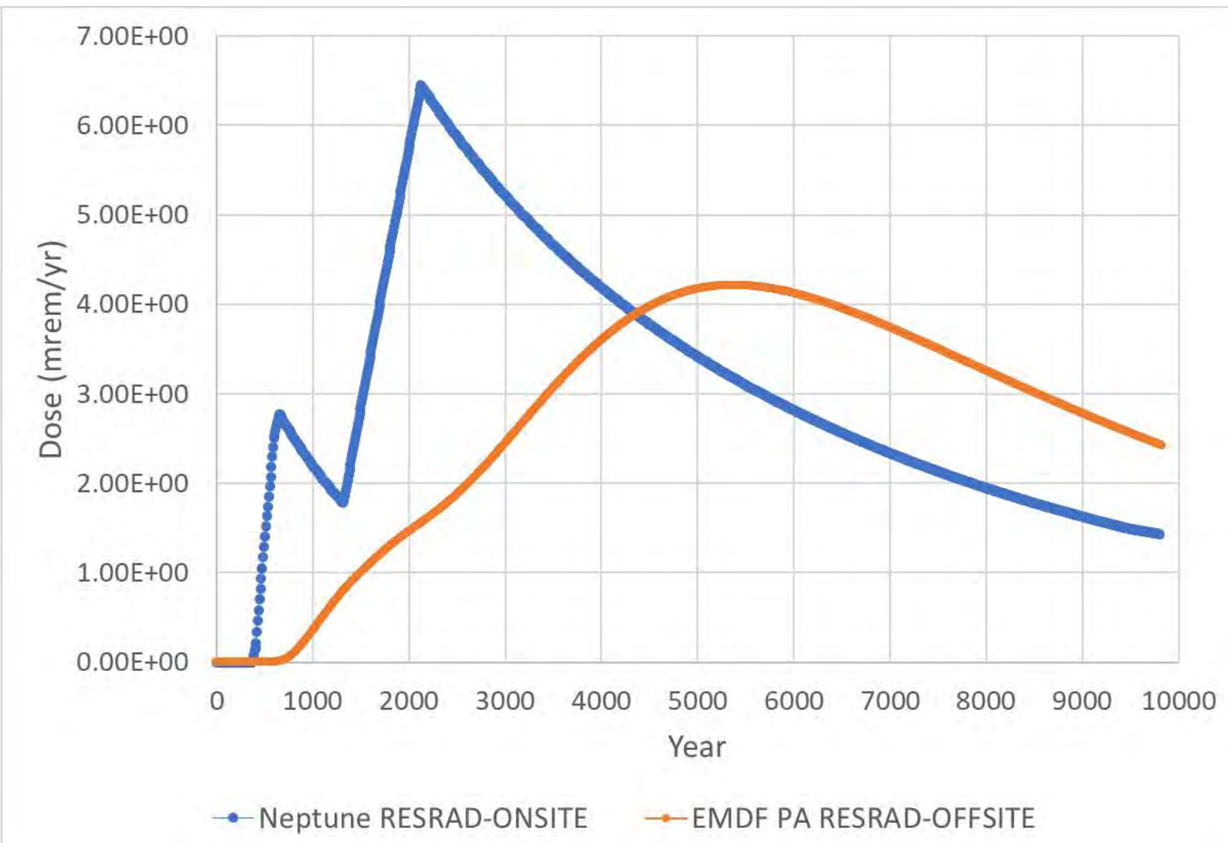

Figure B 3. Comparison of Neptune RESRAD-ONSITE and EMDF PA RESRAD-OFFSITE water-pathways dose over 10,000 years using EMDF PA base case K_d values with year 200 as the time of geomembrane failure.

The divergent results of the EMDF PA base case (RESRAD-OFFSITE) and the RESRAD-ONSITE results are driven by several factors. First, as the program names imply, the receptor scenario for the RESRAD-ONSITE run assumes a water well is located on the EMDF landfill, whereas the RESRAD-OFFSITE scenario assumes the water well is located off-site. Second, Neptune's resident scenario uses radionuclide transfer factors for poultry meat and egg ingestion in place of beef transfer factors, reflecting an emphasis on home-produced foods rather than larger-scale agriculture. Lastly, Neptune obtained parameter values for residence occupancy, intake factors/rates for water, meat (here, chicken and chicken eggs), soil, and other variables from EPA's Exposure Factors Handbook. These values often differ significantly from the values used in the EMDF PA base case scenario. Neptune's RESRAD-ONSITE input values are provided in Section B5.0 of this appendix.

Table B 2. Comparison of Neptune RESRAD-ONSITE and EMDF PA RESRAD-OFFSITE analyses of total water-pathways dose and primary contributing radionuclides for a resident scenario (using year 200 starting concentrations), for the base case K_d condition, over 10,000 years.

Base Case K_d							
Neptune (RESRAD-ONSITE)				***EMDF PA (RESRAD-OFFSITE)***			
Year (post-closure)	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*	*Year (post-closure)*	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*
0	~0	N/A	~0	0	~0	N/A	~0
400	0.16	Tc-99	0.16	400	~0	N/A	~0
500	1.39	Tc-99	0.75	500	1.24×10^{-12}	H-3	1.24×10^{-12}
		C-14	0.64	600	1.81×10^{-11}	Tc-99	1.04×10^{-11}
600	2.55	Tc-99	1.31			H-3	7.58×10^{-12}
		C-14	1.25	1,000	9.82×10^{-2}	Tc-99	7.27×10^{-2}
1,000	2.22	C-14	1.19			C-14	2.55×10^{-2}
		Tc-99	1.03	2,000	1.30	Tc-99	0.85
2,000	4.62	I-129	3.32			C-14	0.41
2,326 (peak)	6.45	I-129	5.14	5,000	4.11	I-129	3.80
		C-14	0.55				
		Tc-99	0.51				
5,000	3.42	I-129	3.27	5,576 (peak)	4.21	I-129	3.98
						Tc-99	0.13
10,000	1.43	I-129	1.40	10,000	2.43	I-129	2.42

B4.2 Drill Cuttings/Garden Exposure Pathways

A component of the PA Resident IHI scenario is exposure to drill cuttings from a water well that might be distributed on the ground surface near and within garden soils. The assumption that cuttings are mixed with garden soils follows the EMDF PA Resident IHI scenario described in Appendix I of the EMDF PA. The concentrations entered into RESRAD to calculate total dose for the "garden" pathways were derived from Year 2047 concentrations listed in PA Appendix I, Table I.1 (EMDF waste average activity concentration) by adjusting to account for hypothetical garden dimensions, calculated according to the method described in Section I.3 of Appendix I (tilling and dilution of contaminated drill cuttings into garden soil). Neptune modified the RESREAD-ONSITE model to evaluate dose related to exposure to drill cuttings in garden soil.

This exposure pathway was evaluated under two K_d scenarios—one using the EMDF PA base case K_d values, the other using the RESRAD DCH K_d values.

As shown in Figure B 4 below, the results of this evaluation suggest that dose peaks at year 0 and quickly decreases from that time as short-lived radionuclides decay. In addition, there is a negligible difference in dose based on choice of K_d values, indicating that leaching of radionuclides does not greatly impact drill cuttings dose over time. The drill cuttings scenario shows calculated doses that would result if drill cuttings from a water well that was drilled at any time (shown on the x-axis of Figure B 4) are mixed into garden soil at that time. Whereas the water-pathways doses shown in Figure B 1 and Figure B 2 result from leaching of soluble radionuclides into groundwater, drill cuttings doses are largely independent of radionuclide mobility. In fact, over long periods of time, as more-soluble radionuclides are depleted from the waste zone, only low-mobility radionuclides (those with relatively large K_d values) remain to contribute to drill cuttings dose.

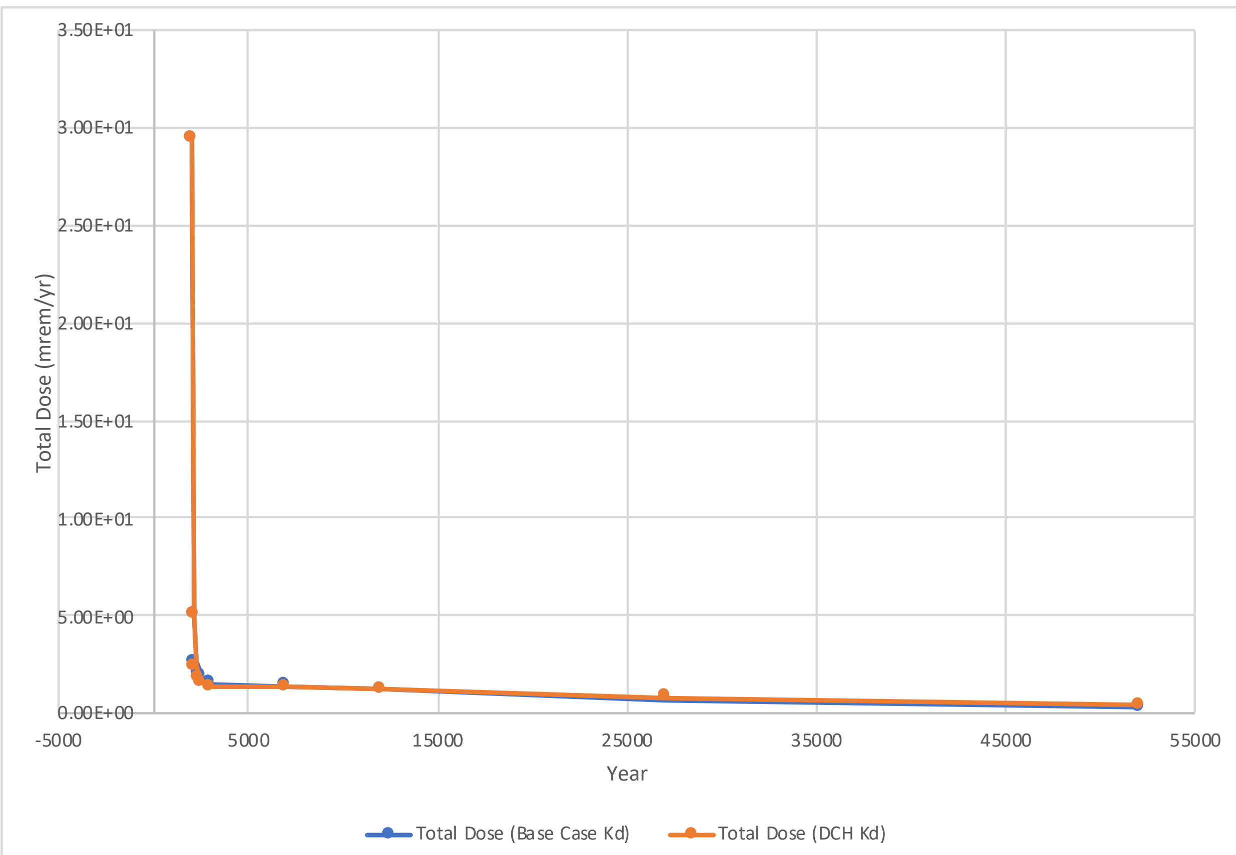

Figure B 4. Total doses to a potential receptor from garden soil contaminated by drill cuttings for two K_d scenarios over ~52,000 years after year 2047.

B4.3 Resident EMDF Receptor Using Natural Recharge Rate

Finally, as a boundary case, Neptune examined a scenario in which, following membrane failure, the site's natural infiltration rate (8 inches per year; PA Section 3.3.1.6) is attained as a result of the liner being compromised from cracking subsequent to differential settling from waste compaction or desiccation during prolonged drought. (Previous model settings described in this evaluation used an infiltration rate of 1 inch per year as a boundary condition.) To test this scenario, Neptune used the estimated EMDF radionuclide concentrations at 300 years (as previously described, these concentrations were derived from ingrowth/decay of radionuclide inventory listed in PA Appendix G, Table G.9), but changed evapotranspiration and runoff coefficient values to achieve an infiltration rate of 8 inches/year (0.203 m/year). The results of RESRAD simulations for this scenario (using two different K_d cases—EMDF PA base case K_d values and RESRAD DCH K_d values) are shown in Figure B 5. Table B 3 contains a detailed breakdown of total dose and important contributing radionuclides at each time point of the different K_d scenarios.

The findings of this model scenario are significant in that they suggest that, in the case of potential membrane and liner failure, if a natural infiltration rate (8 inches/year) is approached, significant total dose and consequent potential risk to human health in the foreseeable future may result if the K_d values are close to those suggested in the RESRAD DCH (Figure B 5). The resulting doses are estimated to peak at approximately 2.7×10^3 millirem per year after approximately 3,300 years, owing mostly to uranic elements from ~2,000 to 5,000 years in the DCH K_d scenario (Table B 3). A similar, but delayed radionuclide profile is observed in the base case K_d scenario, with I-129, C-14, and Tc-99 being the most significant contributors before 5,000 years, at which time uranics begin to dominate, peaking at a dose of 1.6×10^3 millirem per year at approximately year 5,700, nearly 3,000 years after the DCH K_d scenario peak (Table B 3). Importantly, a significant dose on the order of 1,000 mrem/year is estimated at approximately1,700 years in the DCH K_d scenario, while the base case K_d scenario at approximately1,700 years suggests negligible dose, highlighting significant differences between K_d scenarios.

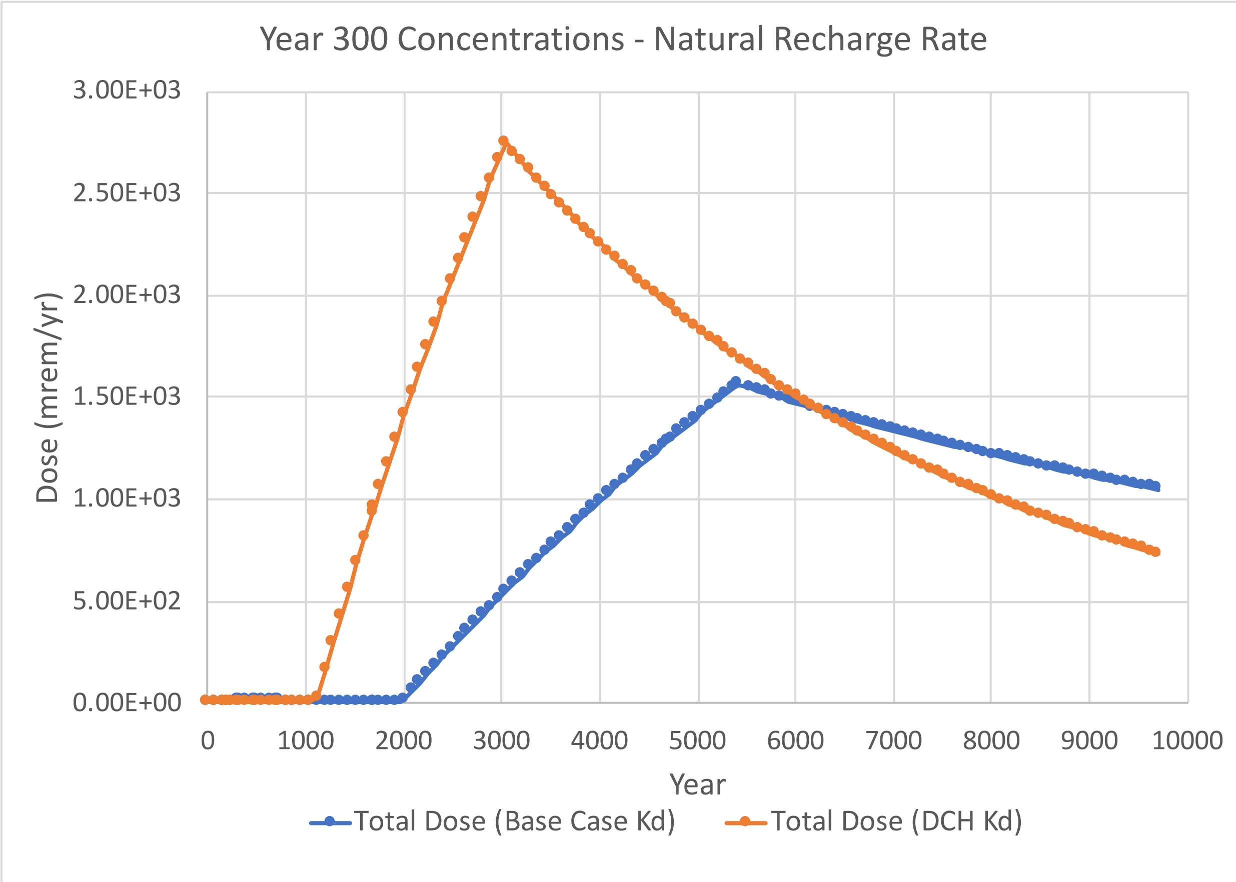

Figure B 5. Total dose to a hypothetical resident on the EMDF landfill using waste concentrations corresponding to geomembrane failure at 300 years and an assumed natural recharge/infiltration rate of 8 inches/year following membrane failure.

Table B 3. Total dose and primary contributing radionuclides in the "resident EMDF receptor using natural recharge rate of 8 inches/year" scenario, for each K_d condition.

Base Case K_d			
Year (post-closure)	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*
300	~0	N/A	~0
500	7.17	C-14	2.91
		Tc-99	2.32
		I-129	1.94
600	10.3	I-129	7.04
		Tc-99	1.41
		C-14	1.09
800	13.9	I-129	12.7
		C-14	0.7
		Tc-99	0.52
1,000	10.0	I-129	9.58
		C-14	0.27
		Tc-99	0.19
2,000	2.74	I-129	2.36
		K-40	0.30
5,000	1.28 x 10^3	U-234	813
		U-238	234
		U-235	150.7
		U-233	55.1
		U-236	11.1
		Pb-210	7.85
5,712 (peak)	1.56 x 10^3	U-234	984.6
		U-238	283
		U-235	182.7
		U-233	66.6
		Pb-210	14.5
10,000	1.05 x 10^3	U-234	593
		U-238	172
		U-235	111
		Pb-210	77.7
		U-233	39.7
		Pa-231	26.1
		U-236	8.15
		Ac-227	7.78
		Th-229	5.91

RESRAD DCH K_d			
Year (post-closure)	*Total Dose (mrem/year)*	*Radionuclide*	*Contributing Dose (mrem/year)*
300	~0	N/A	8~0
500	0.40	Tc-99	0.33
600	0.36	I-129	0.32
800	3.86	I-129	3.86
1,000	6.87	I-129	6.87
2,000	961	U-234	616.2
		U-238	176.1
		U-235	113.5
		U-233	42.0
3,339 (peak)	2.74 x 10^3	U-234	1.76 x 10^3
		U-238	504.6
		U-235	325.4
		U-233	119.8
		U-236	23.9
5,000	1.96 x 10^3	U-234	1.24 x 10^3
		U-238	357.7
		U-235	230.6
		U-233	84.3
		U-236	17.0
		Pb-210	10.9
10,000	732	U-234	436.8
		U-238	126.8
		U-235	81.8
		Pb-210	37.6
		U-233	29.2
		U-236	6.02

B5.0 Documentation of RESRAD-ONSITE Input Parameter Values for the Supplemental Evaluation

This section of Appendix B provides documentation of the input parameter values used in the RESRAD-ONSITE simulations described in Section 4 of this appendix.

B5.1 Radionuclide Decay and Dose Conversion Factors Menu

Neptune used the following sources for radionuclide decay and dose conversion factors (DCFs) inputs:

Radionuclide transformations: ICRP 107 (ICRP 2008) (Table G.5)

Internal dose library: DOE STD-1196-2011, Reference Person (Table G.5)

External dose library: DCFPAK 3.02 (Table G.5)

Cut-off half-life: 180 days (RESRAD-OFFSITE file for Base Case: BASE_180525.ROF)

B5.2 Soil Concentrations Menu

Neptune used base case and model source concentrations from PA Appendix G, Table G.9 as a starting point for evaluation of initial (time zero) radionuclides and corresponding soil source contaminant concentrations. Neptune processed initial concentrations to account for decay and ingrowth in a period of effectively zero-infiltration (zero leaching) conditions during the assumed lifespan of the geomembrane layers (base case = 200 years). These calculations were done in GoldSim version 12.0, with ICRP 107 (ICRP 2008) nuclide data. Appendix G.6 text states: “Geomembrane liners for the EMDF cover and liner systems are expected to be effective in limiting infiltration and controlling releases of leachate for their estimated service life, reported to range from a few hundred years to 1,000 years or more (Koerner et al. 2011, Rowe et al. 2009, Benson 2014).”

RESRAD-ONSITE was run for three assumed effective lifespans (100, 200, and 300 years) to gauge the sensitivity of water-pathways doses to the assumption of geomembrane longevity.

Table B 4. Initial Contaminated Zone Concentrations for Calculation of Water-Pathways Doses; model year zero, model year 200, and model year 300.

Isotope	Source Conc, 0 yr (pCi/g)	Source Conc, 100 yr (pCi/g)	Source Conc, 200 yr (pCi/g)	Source Conc, 300 yr (pCi/g)
Ac-227	2.17E-03	4.08E-01	6.69E-01	9.23E-01
Am-241	5.50E+01	5.28E+01	4.50E+01	3.83E+01
Am-243	5.44E+00	5.39E+00	5.34E+00	5.29E+00
Be-10	4.67E-05	4.67E-05	4.67E-05	4.67E-05
C-14	2.93E+00	2.89E+00	2.86E+00	2.83E+00
Ca-41	4.21E-02	4.21E-02	4.20E-02	4.20E-02

Isotope	Source Conc, 0 yr (pCi/g)	Source Conc, 100 yr (pCi/g)	Source Conc, 200 yr (pCi/g)	Source Conc, 300 yr (pCi/g)
Cm-243	3.81E-01	3.52E-02	3.26E-03	3.01E-04
Cm-244	1.11E+02	2.42E+00	5.26E-02	1.14E-03
Cm-245	3.33E-02	3.30E-02	3.28E-02	3.25E-02
Cm-246	1.59E-01	1.57E-01	1.54E-01	1.52E-01
Cm-247	7.80E-03	7.80E-03	7.80E-03	7.80E-03
Cm-248	1.11E-02	1.11E-02	1.11E-02	1.11E-02
Cs-137	1.24E+03	1.25E+02	1.25E+01	1.26E+00
H-3	1.04E+01	3.76E-02	1.36E-04	4.92E-07
I-129	3.40E-01	3.40E-01	3.40E-01	3.40E-01
K-40	2.84E+00	2.84E+00	2.84E+00	2.84E+00
Mo-93	3.88E-01	3.81E-01	3.75E-01	3.68E-01
Nb-93m	2.33E-01	8.16E-01	8.18E-01	8.13E-01
Nb-94	9.63E-01	9.60E-01	9.56E-01	9.53E-01
Ni-59	3.04E+00	3.04E+00	3.04E+00	3.03E+00
Np-237	3.30E-01	3.32E-01	3.33E-01	3.35E-01
Pa-231	2.39E-01	4.94E-01	7.49E-01	1.00E+00
Pb-210	3.19E+00	8.56E-01	8.19E-01	9.03E-01
Pd-107	1.41E-02	1.41E-02	1.41E-02	1.41E-02
Pu-238	8.38E+01	3.80E+01	1.73E+01	7.83E+00
Pu-239	5.41E+01	5.40E+01	5.38E+01	5.37E+01
Pu-240	5.77E+01	5.74E+01	5.68E+01	5.62E+01
Pu-241	2.04E+02	1.67E+00	4.59E-02	3.27E-02
Pu-242	1.72E-01	1.72E-01	1.72E-01	1.72E-01
Pu-244	3.68E-03	3.68E-03	3.68E-03	3.68E-03
Ra-226	7.12E-01	7.64E-01	8.39E-01	9.35E-01
Ra-228	2.17E-02	3.38E+00	3.38E+00	3.38E+00
Se-79	9.62E-02	9.62E-02	9.62E-02	9.61E-02
Sn-121m	2.22E+01	4.58E+00	9.44E-01	1.95E-01
Sn-126	1.52E-01	1.52E-01	1.52E-01	1.52E-01
Sr-90	3.94E+02	3.55E+01	3.20E+00	2.88E-01
Tc-99	2.97E+00	2.97E+00	2.97E+00	2.97E+00
Th-228	1.21E-06	7.22E+00	4.78E+00	3.89E+00
Th-229	5.71E+00	6.05E+00	6.38E+00	6.71E+00
Th-230	1.66E+00	2.24E+00	2.81E+00	3.38E+00
Th-232	3.38E+00	3.38E+00	3.38E+00	3.38E+00
U-232	1.02E+01	3.73E+00	1.36E+00	4.99E-01
U-233	4.16E+01	4.16E+01	4.16E+01	4.15E+01
U-234	6.28E+02	6.28E+02	6.28E+02	6.28E+02
U-235	1.21E+02	1.21E+02	1.21E+02	1.21E+02
U-236	9.00E+00	9.00E+00	9.00E+00	9.00E+00
U-238	1.82E+02	1.82E+02	1.82E+02	1.82E+02
Zr-93	5.00E-01	5.00E-01	5.00E-01	5.00E-01

Neptune also used RESRAD-ONSITE to evaluate dose in a scenario in which drill cuttings from a water well have been mixed with garden soil; this evaluation served as a comparison to the Resident inadvertent human intruder (IHI) scenario presented in Appendix I of the EMDF PA, which was limited in scope. Lastly, Neptune used RESRAD-ONSITE to evaluate the bounding dose case of exposure of a hypothetical receptor on the EMDF landfill at a time when the engineered cover has been assumed to be completely eroded. Calculation of these source concentrations is described below.

Table B 5. Initial Contaminated Zone Concentrations for Calculation of Garden Drill Cuttings Doses, and Doses to a Hypothetical Receptor Residing on the Landfill Subsequent to Erosion of the Cover.

Isotope	Source Conc, Garden (pCi/g)	Source Conc, Waste Mixed with Fill (pCi/g)
Ac-227	1.40E-05	2.17E-03
Am-241	4.80E-01	7.43E+01
Am-243	3.50E-02	5.42E+00
C-14	1.89E-02	2.93E+00
Cd-113m	2.77E-03	4.29E-01
Cf-249	6.93E-09	1.07E-06
Cf-250	4.77E-08	7.38E-06
Cf-251	1.36E-09	2.10E-07
Cf-252	6.79E-08	1.05E-05
Cm-243	2.46E-03	3.81E-01
Cm-244	7.13E-01	1.10E+02
Cm-245	2.15E-04	3.32E-02
Cm-246	1.03E-03	1.59E-01
Cm-247	5.04E-05	7.81E-03
Cm-248	7.17E-05	1.11E-02
Co-60	1.45E-04	2.25E-02
Cs-134	4.60E-11	7.12E-09
Cs-135	2.72E-03	4.21E-01
Cs-137	8.03E+00	1.24E+03
Eu-152	1.72E-01	2.67E+01
Eu-154	3.70E-02	5.73E+00
Eu-155	3.25E-04	5.04E-02
Fe-55	3.98E-09	6.16E-07
H-3	6.69E-02	1.04E+01
I-129	2.20E-03	3.40E-01
K-40	1.83E-02	2.84E+00
Kr-85	6.00E-01	9.29E+01
Mo-100	2.71E-08	4.20E-06
Na-22	5.25E-09	8.12E-07
Nb-94	6.21E-03	9.61E-01
Ni-59	1.97E-02	3.04E+00

Isotope	Source Conc, Garden (pCi/g)	Source Conc, Waste Mixed with Fill (pCi/g)
Ni-63	2.33E+00	3.61E+02
Np-237	2.13E-03	3.30E-01
Pa-231	1.54E-03	2.38E-01
Pb-210	2.06E-02	3.19E+00
Pd-107	9.12E-05	1.41E-02
Pm-146	1.45E-06	2.25E-04
Pm-147	8.06E-05	1.25E-02
Pu-238	5.42E-01	8.39E+01
Pu-239	3.50E-01	5.42E+01
Pu-240	3.74E-01	5.79E+01
Pu-241	1.32E+00	2.04E+02
Pu-242	1.11E-03	1.72E-01
Pu-244	2.38E-05	3.68E-03
Ra-226	4.60E-03	7.12E-01
Ra-228	1.40E-04	2.17E-02
Re-187	1.10E-08	1.70E-06
Sb-125	1.54E-06	2.38E-04
Se-79	6.21E-04	9.61E-02
Sm-151	2.71E+00	4.19E+02
Sn-126	9.81E-04	1.52E-01
Sr-90	2.54E+00	3.94E+02
Tc-99	1.92E-02	2.97E+00
Th-228	7.79E-09	1.21E-06
Th-229	3.67E-02	5.68E+00
Th-230	1.07E-02	1.66E+00
Th-232	2.18E-02	3.38E+00
U-232	6.59E-02	1.02E+01
U-233	2.69E-01	4.16E+01
U-234	4.05E+00	6.27E+02
U-235	7.79E-01	1.21E+02
U-236	5.83E-02	9.03E+00
U-238	1.18E+00	1.82E+02
Zr-93	3.23E-03	5.00E-01

B5.3 K_d and Transfer Factors Menus

Neptune evaluated two EMDF scenarios to account for two different partition (distribution) coefficient (K_d) input lists. The first case used element-specific “base case” K_d values from PA Appendix G, Table G.11, while the second case was evaluated using recommended K_d values (geometric mean) for clay soil type (most analogous to shale) from the RESRAD *Data Collection Handbook* (DCH), Table 2.13.3 (Yu et al. 2015).

Default plant transfer factors ($K_{p\text{-}s}$) in RESRAD-ONSITE v7.2 were applied. These are identical to those used in the RESRAD-OFFSITE "base case" calculations, because the "Library" of dose factors, slope factors, and transfer factors is common to both programs.

Transfer factors for chicken meat (TF_{chk}) and eggs (TF_{egg}) are used to evaluate radionuclide exposure from domestic livestock. These animals are more applicable to a rural resident exposure scenario than cows or cattle. RESRAD-ONSITE allows for a single meat transfer factor, so a weighted factor (TF_{wtd}) is calculated from TF_{chk} and TF_{egg} using the relative ingestion rates for chicken meat and eggs. 75th percentile values for intake of home-produced poultry (2.19 g/kg-day; EPA (2011), Table 13-52) and intake of home-produced eggs (0.90 g/kg-day; EPA (2011), Table 13-40) were used to calculate TF_{wtd}. 75th percentile values provide a measure of protectiveness, but avoid the much larger values that can occur on the tails of a distribution based on limited sample size. Selecting all upper-bound values for multiple food pathways (in this case, garden produce, chicken, and eggs) can result in unrealistically large combined quantities of home-grown foods. Values of TF_{wtd} were calculated according to:

$$TF_{wtd} = [\ TF_{chk} \times 1.54 / (1.54 + 0.90)\ ] + [\ TF_{egg} \times 0.90 / (1.54 + 0.90)\ ]$$

The primary sources for chicken and chicken egg transfer factors were taken from the International Atomic Energy Agency (IAEA's) *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments* (IAEA 2010) and *A Compendium of Transfer Factors for Agricultural and Animal Products* (Staven et al. 2003). IAEA (2010) included information on seven nuclides to inform distributions: Cd, Co, Cs, I, Se, Sr, and U. The remaining elements were assigned to a generic distribution based on their element group using lognormal distributions provided for each nuclide in IAEA (2010) and point estimates provided for each nuclide in Staven et al. (2003). Geometric means and geometric standard deviations were calculated for elements which were assigned lognormal distributions. Due to a lack of unique values, a TF generic distribution was not estimable for the following element groups: tetrels, triels, titanium group, and vanadium group. As a result, the average geometric standard deviation reported in IAEA (2010) among the poultry data was assigned to one record per element group to estimate a generic distribution (Sn in tetrels, Tl in triels, Zr in titanium group, and Nb in vanadium group). The values for Ca and K are taken from a single value in NUREG/CR5512.

RESRAD-ONSITE was run using EMDF PA and DCH K_d inputs to gauge model sensitivity to these values.

Table B 6. Element-specific values for K_d, $K_{p\text{-}s}$, TF_{chk} and TF_{egg}.

Element	K_d, PA Table G.11 (ml/g)	K_d, DCH Table 2.13.3 (ml/g)	Default $K_{p\text{-}s}$, DCH Table 6.3.10	TF_{chk} (pCi/kg)/(pCi/d)	TF_{egg} (pCi/kg)/(pCi/d)	TF_{wtd} (pCi/kg)/(pCi/d)
Ac	1,500	2,400	0.0025	0.27	0.012	0.175
Am	1,000	8,100	0.001	0.27	0.012	0.175
Be	810	1,300	0.004	0.026	0.36	0.149
C	1.09	0 (b)	5.5	2.60	1.95	2.36

Element	K_d, PA Table G.11 (ml/g)	K_d, DCH Table 2.13.3 (ml/g)	Default K_{p-s}, DCH Table 6.3.10	TF_{chk} (pCi/kg)/(pCi/d)	TF_{egg} (pCi/kg)/(pCi/d)	TF_{wtd} (pCi/kg)/(pCi/d)
Ca	50	16	0.5	0.044	0.44	0.190
Cm	1,000	5,400	0.001	0.27	0.012	0.175
Cs	3,000	5,500	0.04	2.7	0.40	1.85
H	0.199	0 (b)	4.8	4.12	4.53	4.27
I	4	7	0.02	0.0087	2.4	0.891
K	30	43	0.3	0.40	0.700	0.511
Mo	125	90	0.13	0.19	0.64	0.356
Nb	500	2,400	0.01	0.00028	0.0011	0.000582
Ni	500	930	0.05	0.00055	0.020	0.00772
Np	40	36 (a)	0.02	0.27	0.012	0.175
Pa	400	2,700	0.01	0.042	0.012	0.0309
Pb	1,000	2,100 (a)	0.01	0.78	0.98	0.854
Pd	180	270	0.1	0.00055	0.020	0.00772
Pu	400	1800	0.001	0.27	0.00015	0.170
Ra	3,000	13,000	0.04	0.026	0.36	0.149
Se	500	240	0.1	9.7	16.0	12.0
Sn	450	670	0.0025	0.78	0.98	0.854
Sr	30	95	0.3	0.020	0.35	0.142
Tc	1	0.09	5.0	0.0079	0.13	0.0529
Th	3,000	4,500	0.001	0.27	0.012	0.175
U	50	28	0.0025	0.600	1.1	0.784
Zr	50	410 (a)	0.001	0.000053	0.00020	0.000107
(a) A geometric mean was not available, the geometric mean for generic soil (Table 2.13.5) was used. (b) Values unavailable in Tables 2.13.3 and 2.13.5, RESRAD default value applied.						

B5.4 Contaminated Zone Menu

For the "Contaminated Zone Menu" spatial inputs at the EMDF site, Neptune used the following values and sources:

Area of contaminated zone (CZ): 92,590 m^2 (PA Appendix G, Attachment G.1)

Thickness of CZ: 18.288 m (PA Appendix G, Attachment G.1; average thickness)

Length parallel to aquifer flow[1]: 382 m (PA Appendix G, Attachment G.1)

[1] The distance between two parallel lines perpendicular to the direction of aquifer flow, one at the upgradient edge of the contaminated zone and the other at the downgradient edge of the contaminated zone.

B5.5 Cover/Hydrology Menu

For the "Cover/Hydrology Menu" spatial inputs, Neptune used the following values and sources:

Cover depth: 3.353 m (PA Appendix G, Attachment G.1; EMDF design)

Cover dry bulk density: 1.5 g/cm^3 (PA Appendix G, Attachment G.1; site soil characteristics)

Cover (and CZ) erosion rate: 6.6E^{-05} m/year (PA Appendix C, Table C.7, base case)

CZ total porosity: 0.419 (PA Appendix G, Attachment G.1; HELP model material property)

CZ field capacity: 0.307 (PA Appendix G, Attachment G.1; HELP model material property)

CZ hydraulic conductivity: 5.99 m/year (PA Appendix G, Attachment G.1; HELP model material property)

CZ b parameter: 5.30 (PA Appendix G, Attachment G.1; HELP model material property)

Humidity in air: 8 g/m^3 (PA Appendix G, Attachment G.2; Base Case RESRAD-OFFSITE summary report)

Wind speed: 3.4342 m/second (PA Appendix G, Attachment G.1; site-specific data)

Parameters related to the "degraded condition" long-term infiltration rate of 1inch per year (0.0254 m/year). A realistic irrigation rate is used because irrigation is the only mechanism by which soil and garden crops are impacted by radionuclides leaching from EMDF wastes.

from the RESRAD 6 Manual, Eq E.4: $I = (1 - C_e)[(1 - C_r)P_r + I_{irr}]$

where

C_e = evapotranspiration coefficient
C_r = runoff coefficient
P_r = precipitation rate
I_{irr} = irrigation rate

Evapotranspiration coefficient: 0.91 (value used to return an infiltration rate of 0.0254 m/year)

Runoff coefficient: 0.904 (value used to return an infiltration rate of 0.0254 m/year)

Precipitation rate: 1.382 m/year (PA Appendix G, Attachment G.1; site-specific data)

Irrigation rate: 0.15 m (PA Appendix G, Attachment G.1; all crops, referenced to USDA 2014 - Table 4. Water Resources Region 6 Tennessee [0.5 acre-ft per acre, on-farm surface water])

Parameters contributing to the calculation of the infiltration rate were also defined to represent a condition where the clay layer in the cover has failed, such as might occur due to cracking subsequent to differential settling from waste compaction or desiccation during a prolonged drought. An infiltration of 8″ per year (0.203 m/year) was used to represent a natural recharge rate for this condition (Section 3.3.1.6 of the PA Report). The values of the evapotranspiration and runoff coefficients were changed to return an infiltration rate of 0.203 m/year:

Evapotranspiration coefficient: 0.665 (value used to return an infiltration rate of 0.203 m/year)

Runoff coefficient: 0.67 (value used to return an infiltration rate of 0.203 m/year)

B5.6 Saturated Zone Menu

For the RESRAD-ONSITE inputs relating to the saturated zone, Neptune used the following values and sources:

Saturated zone dry bulk density: 1.8 g/cm^3 (PA Appendix G, Table G.15)

Saturated zone total porosity: 0.35 (PA Appendix G, Table G.15)

Saturated zone effective porosity: 0.27 (PA Appendix G, Table G.15)

Saturated zone hydraulic conductivity: 83.6 m/year (PA Appendix G, Table G.15)

Saturated zone field capacity: 0.247 (PA Appendix G, Table G.14; UZ5 value [bottom-most unsaturated zone, above the water table], this parameter is not used in RESRAD-OFFSITE, so no EMDF PA reference is available)

Saturated zone hydraulic gradient: 0.033 (PA Appendix G, Table G.15; MODFLOW model results)

Water table drop rate: 0 m/year (This parameter is not used in RESRAD-OFFSITE, a value of zero implies an unchanging water table throughout the model period)

Well pump intake depth: 10 m below water table (RESRAD-ONSITE default)

Well pumping rate: 250 m^3/year (RESRAD-ONSITE default)

B5.7 Unsaturated Zone Menu

For the RESRAD-ONSITE inputs relating to the unsaturated zone, Neptune used the following values from PA Appendix G, Table G.14:

Table G.14. Unsaturated zone hydrology

Zone number	Thickness (m)	Dry bulk density (g/cm^3)	Total porosity	Effective porosity	Field capacity	Hydraulic conductivity (m/year)	b parameter
UZ1	0.305	1.4	0.463	0.294	0.232	117	5.4
UZ2	0.305	1.6	0.397	0.389	0.032	94600	4.05
UZ3	0.914	1.5	0.427	0.195	0.418	0.315	11.4
UZ4	3.05	1.5	0.445	0.236	0.393	0.599	11.4
UZ5	3.05	1.8	0.353	0.270	0.247	11.1	10.4

B5.8 Occupancy Menu

The following values and sources were used by Neptune for RESRAD-ONSITE inputs relating to resident occupancy:

Inhalation rate: 5,303 m^3/year

Mass loading for inhalation: 1.21E-06 g/m^3 (Calculated using EPA's Particulate Emission Factor model, described in EPA's *Supplemental Guidance for Developing Soil Screening Levels for Superfund Sites* [December 2002], assuming atmospheric conditions for Atlanta, GA [Exhibit D-2].)

Exposure Duration: 30 years (RESRAD-ONSITE default; not applicable to an annual dose endpoint)

Indoor dust filtration factor: 0.4 (RESRAD-ONSITE default)

External gamma shielding factor: 0.7 (RESRAD-ONSITE default)

Indoor time fraction: 0.666

Outdoor time fraction: 0.0918

Shape of contaminated zone: circular

The inhalation rate was calculated using the mean daily inhalation rate weighted for ages 0 to <81 years (14.52 m^3/day) from Table 6-1 of the *Exposure Factors Handbook* (EPA 2011), and then multiplying by the number of days in a year (365.25) to derive a yearly inhalation rate of 5303 m^3/year.

The indoor and outdoor time fractions were calculated using the mean daily time spent indoors and outdoors at the residence across all populations from EPA (2011) Tables 16-16 (1,001 min/day) and 16-20 (138 min/day), respectively. Full calculations were for yearly time fractions are as follows, using the EPA guidance for days per year at the home residence:

$$\text{Yearly indoor time fraction: } \frac{1001\,\frac{min}{day} \times 24\,\frac{hr}{day}}{1440\,\frac{min}{day}} = \frac{16.68\,\frac{hr}{day} \times 350\,\frac{days}{year}}{8766\,\frac{hrs}{year}} = \mathbf{0.666}$$

Yearly outdoor time fraction: $\frac{138\,\frac{min}{day} \times 24\,\frac{hr}{day}}{1440\,\frac{min}{day}} = \frac{2.3\,\frac{hr}{day} \times 350\,\frac{days}{year}}{8766\,\frac{hrs}{year}} = \mathbf{0.0918}$

For inhalation rate and indoor/outdoor time fractions, mean values averaged across all ages and populations from the EFH were chosen because they best approximate the hypothetical reference person employed by ICRP to develop the DCFs used in the RESRAD-ONSITE model described in this memo.

B5.9 Ingestion Pathway, Dietary Menu

The following values and sources were used and/or calculated by Neptune for RESRAD-ONSITE inputs relating to the dietary practices of an on-site resident. Explanation of the calculated values are also provided below.

Fruit, vegetable, and grain consumption: 97.3 kg/year

Leafy vegetable consumption: 0 kg/year

Meat and poultry consumption: 74.29 kg/year

Soil ingestion: 16.46 g/year

Drinking water intake: 409.1 L/year

Contaminated Fractions:

- Drinking water: 1
- Livestock water: 1
- Irrigation water: 1
- Plant food: 1
- Meat: 1

B5.9.1 Calculation of Dietary Ingestion Pathway Values

Fruit, vegetable, and grain ingestion parameters were calculated using total 75th-percentile daily consumer-only home-produced fruit and vegetable ingestion rates for all ages and regions from Table 13-30 (2.35 g/kg-day and 1.80 g/kg-day) in EPA (2011). These values were then adjusted to account for net preparation and cooking losses using the percentages from EPA (2011) Table 13-69 for fruits and vegetables (25.4% and 12.4%, respectively). Next, seasonally- and preparation/cooking loss-adjusted ingestion rates were extrapolated to yearly values based on an 80-kg person to derive the final annual consumption rates, which were then combined for the final parameter value typed in the dietary menu of RESRAD-ONSITE. Note that home-milled grain was considered to be outside the scope of the casual home gardener/farmer, and thus, only home-produced fruit and vegetable consumption is considered for the "Fruit, vegetable, and grain consumption" parameter.

Additionally, leafy vegetable intake is included, or "rolled up," in the "fruit, vegetable, and grain consumption" parameter and thus a value of zero was used for the leafy vegetable consumption input in this model. Consequently, this means that only root uptake is used to model leafy vegetable concentrations.

Annual home-produced fruit consumption: $\frac{2.35\,\frac{g}{kg-day} \times 0.746\,net\,\% \times 80\,kg\,BW \times 365.25\,\frac{days}{year}}{1000\,\frac{g}{kg}} = 51.23\,\frac{kg}{year}$

Annual home-produced vegetable consumption: $\frac{1.80\,\frac{g}{kg-day} \times 0.876\,net\,\% \times 80\,kg\,BW \times 365.25\,\frac{days}{year}}{1000\,\frac{g}{kg}} = 46.07\,\frac{kg}{year}$

Annual home-produced fruit and vegetable consumption: $51.23\,\frac{kg}{year} + 46.07\,\frac{kg}{year} = \mathbf{97.3}\,\frac{\mathbf{kg}}{\mathbf{year}}$

Likewise, meat and poultry consumption (in this residential scenario, this value represents the combined consumption of poultry and poultry eggs) was calculated using total 75th percentile daily consumer-only home-produced poultry and egg ingestion rates for all populations from Tables 13-52 (2.19 g/kg-day) and 13-40 (0.90 g/kg/-day) in EPA (2011). Poultry intake was then adjusted to account for net preparation and cooking losses using the percentage listed in EPA (2011) Table 13-69 for meat (29.7%); eggs were deemed to not lose any weight other than egg shells during cooking and therefore, the egg intake rate was not adjusted. Next, preparation/cooking loss-adjusted ingestion rates were extrapolated to yearly values based on an 80-kg person to derive the final annual consumption rates for poultry and eggs, which were then combined for the final parameter value for "meat and poultry consumption" typed in the dietary menu of RESRAD-ONSITE.

Annual home-produced poultry consumption: $\frac{2.19\,\frac{g}{kg-day} \times 0.703 \times 80\,kg\,BW \times 365.25\,\frac{days}{year}}{1000\,\frac{g}{kg}} = 44.99\,\frac{kg}{year}$

Annual home-produced eggs consumption: $\frac{0.90\,\frac{g}{kg-day} \times 80\,kg\,BW \times 365.25\,\frac{days}{year}}{1000\,\frac{g}{kg}} = 29.3\,\frac{kg}{year}$

Annual home-produced poultry and egg consumption: $44.99\,\frac{kg}{year} + 29.3\,\frac{kg}{year} = \mathbf{74.29}\,\frac{\mathbf{kg}}{\mathbf{year}}$

Soil ingestion rate was calculated using the general population central tendency soil and dust ingestion values from EPA (2017) (Soil + Dust, Table 5-1, see column highlighted in table below; this ingestion rate accounts for soil ingestion and indoor settled dust).

Table 5-1. Recommended Values for Daily Soil, Dust, and Soil + Dust Ingestion (mg/day)[a]

Age Group	Soil + Dust		Soil[b]				Dust[c]	
	General Population Central Tendency[d]	General Population Upper Percentile[e]	General Population Central Tendency[f]	General Population Upper Percentile[f]	Soil Pica[g]	Geophagy[h]	General Population Central Tendency[f]	General Population Upper Percentile[f]
<6 months	40	100	20	50	—	—	20	60
6 months to <1 year	70 (60–80)	200	30	90	—	—	40	100
1 to <2 years	90	200	40	90	1,000	50,000	50	100
2 to <6 years	60	200	30	90	1,000	50,000	30	100
1 to <6 years	80 (60–100)	200	40	90	1,000	50,000	40	100
6 to <12 years	60 (60–60)[i]	200	30	90	1,000	50,000	30	100
12 years through adult	30 (4–50)[j]	100[j]	10	50	—	50,000	20	60

These rates were normalized to a lifetime of 80 years to derive an average daily soil consumption rate of 35.63 mg/day (i.e. 0.03563 g/day), using the following methodology. First, 1 was divided by the number of months in 80 years (960 months) to determine the contributing fraction for each month (0.00104/month). Next, the relevant soil ingestion rate for each age group was multiplied by the fraction of 960 months represented by the age group. For example, for the <6 months age group, 40 mg/day was multiplied by the age group fractional contribution to total time (960 months) of 0.00624 (0.00104/month × 6 months), resulting in the total contribution of the <6 months age group to the final soil ingestion rate being 0.2496 mg/day. This calculation was repeated by each contributing age group (note: the 1 to <6 years age group soil ingestion rate [80 mg/day] was used in place of separate soil ingestion rates for the 1 to < years and 2 to <6 years age groups).

The final annual soil ingestion rate was calculated to compensate for the time-based occupancy factor applied by RESRAD in calculating exposure from the soil ingestion pathway as shown below:

$$\text{Annual soil ingestion rate: } \frac{Soil\ ing.rate\left(\frac{mg}{day}\right)\times Total\ occupancy\left(\frac{days}{year}\right)}{1000\,\frac{mg}{g}\times(Time\ Frac.\ Indoor+Time\ Frac.\ Outdoor)} = \frac{35.63\,\frac{mg}{day}\times 350\,\frac{days}{year}}{1000\,\frac{mg}{g}\times(0.666+0.0918)} = \mathbf{16.46}\ \frac{\boldsymbol{g}}{\boldsymbol{year}}$$

The annual drinking water ingestion rate was calculated based on the mean daily per capita drinking water ingestion rates for all ages in Table 3-1 of the *Exposure Factors Handbook* (14 mL/kg-day; EPA (2011)). This value was then adjusted to account for a body weight of 80 kg and extrapolated to an annual rate for a final value of 409.1 L/year.

$$\text{Annual drinking water ingestion rate: } \frac{Water\ ing.\ rate\left(\frac{mL}{kg-day}\right)\times Body\ weight\ (kg)\times average\ \ days\ per\ year}{1000\,\frac{mL}{L}} =$$

$$\frac{14\,\frac{mL}{kg-day}\times 80\ kg\times 365.25\,\frac{days}{year}}{1000\,\frac{mL}{L}} = \mathbf{409.1}\ \frac{\boldsymbol{L}}{\boldsymbol{year}}$$

B5.9.2 Use of 75th Percentile Values for Produce, Chicken, and Egg Intake Rates

75th percentile values provide a measure of protectiveness, but avoid the much larger values that can occur on the tails of a distribution based on limited sample size. Selecting all upper-bound values for multiple food pathways (in this case, garden produce, chicken, and eggs) can result in unrealistically large combined quantities of home-grown foods.

B5.9.3 Body Weight Normalization

Neptune acknowledges that the use of 80 kg to represent a lifetime body weight in the food and soil ingestion equations above may underestimate the true age-adjusted/normalized consumption rates (because over a lifetime, the average body weight is likely less than 80 kg when considering lower weights early and late in life). However, this is a standard weight for a reference adult in EPA guidance (Table 8-1, EPA (2011)), and calculating ingestion rates at different life stages for fruit/vegetable and chicken/egg consumption is inhibited by a lack of data for early life stage consumption rates in the data presented as part of the *Exposure Factors Handbook* (EPA 2011). Additionally, for this model, the 75th percentile values for fruit/vegetable and chicken/egg consumption rates were taken. Using an 80-kg reference body weight will, if anything, underestimate annual consumption rates, and thus, will lessen any further conservative bias in dose calculations.

B5.10 Ingestion Pathway, Non-dietary Menu

The following values and sources were used and/or calculated by Neptune for RESRAD-ONSITE inputs relating to the non-dietary (i.e. non-human dietary habits) ingestion values relevant to an on-site resident.

Livestock fodder intake for meat: 0.08 kg/day

Livestock water intake for meat: 0.14 L/day

Livestock intake of soil: 0.011 kg/day

Mass loading for foliar deposition: 0.0001 g/m^3 (RESRAD-ONSITE default; unlike the general mass loading value in the Occupancy menu, garden mass loading is assumed to include contributions from mechanical disturbances)

Depth of soil mixing layer: 0.15 m (RESRAD-ONSITE default)

Depth of roots: 0.15 m (Root zone of garden produce assumed to be primarily in the tilled mixing layer)

Here, "livestock" refers to poultry, rather than beef cattle or milk cows. As described below, values for water, soil, and fodder (plant) intake for chickens are used in these calculations. Average fodder intake was estimated to be 80 g/day, water intake 140 mL/day, and intake of soil 11 g/day. Chickens raised for eggs (Leghorn chickens) or meat production (broilers) were not distinguished from each other and both were used to estimate average plant matter and water ingestion rates. These ingestion rates were informed by data from the National Research Council (1994), including weekly feed and water consumption rates for brown-egg-laying and white-egg-laying leghorn-type chickens, aged 0–20 weeks (aged 1–20 weeks for the water consumption data), and broiler chickens aged 1–9 weeks (aged 1–8 weeks for the water consumption data). The average chicken soil ingestion rate is informed by data from Waegeneers et al. (2009). In particular, Neptune used their estimated soil intakes when surface area per chicken is >50 m^2.

B5.10.1 Groundwater Fractional Usage (Balance from Surface Water)

- Drinking water: 1
- Livestock water: 1
- Irrigation water: 1

B5.11 Radon Menu

The Radon Menu in RESRAD-ONSITE was switched off because we did not evaluate the intricacies of radon infiltration/flux at the site.

B5.12 Storage Times Menu

The following values and sources were used by Neptune for RESRAD-ONSITE inputs relating to the storage time of various foods for an on-site resident. “Not used” means the parameter was switched off in the RESRAD model.

Fruits, non-leafy vegetables, and grain: 1 day (most garden produce assumed to be eaten close to time it is harvested)

Leafy vegetables: 1 day (RESRAD-ONSITE default; not used)

Meat: 3 days (most meat and eggs assumed to be eaten close to time of butchering and collection of eggs)

Well water: 1 day (RESRAD-ONSITE default)

Surface water: 1 day (RESRAD-ONSITE default; not used)

Livestock fodder: 1 day (chicken fodder is assumed to be foraged or grown on-site)

B5.13 C-14 Menu

The following values and sources were used by Neptune for RESRAD-ONSITE inputs relating to stable (C-12) and radioactive (C-14) carbon concentration and mobility at the hypothetical site.

C-12 concentration in local water: 0.00002 g/cm^3 (RESRAD-ONSITE default)

C-12 concentration in contaminated soil: 0.03 g/g (RESRAD-ONSITE default)

Fraction of vegetation carbon absorbed from soil: 0.02 (RESRAD-ONSITE default)

Fraction of vegetation carbon absorbed from air: 0.98 (RESRAD-ONSITE default)

Thickness of evasion layer of C-14 in soil: 0.3 m (RESRAD-ONSITE default)

C-14 evasion flux rate from soil: 3.8E-07 per second (Evasion rate for clay soil type is 12/year in RESRAD 6 Manual, Table L.2)

C-12 evasion flux rate from soil: 3.8E-07 per second (Evasion rate for clay soil type is 12/year in RESRAD 6 Manual, Table L.2; the transport of C-14 is assumed to follow that of stable carbon, i.e. C-12, in the environment)

B5.13.1 Grain Fraction in Livestock Feed (Balance is Hay/Fodder)

- Beef cattle: 0 (Chickens are assumed to be free-range and any supplemental feed is assumed to consist of plant material grown on-site)
- Milk cow: 0 (Chickens are assumed to be free-range and any supplemental feed is assumed to consist of plant material grown on-site)

B5.14 Revision to RESRAD-ONSITE Parameter Values for Evaluating Chronic Post-Drilling Resident Exposure to Drill Cuttings Mixed into Garden Soil

The RESRAD-ONSITE model used to evaluate water-pathways exposure for the Chronic Post-Drilling Resident scenario was modified to evaluate dose related to exposure to drill cuttings in garden soil. The following calculations were made to create the RESRAD-ONSITE contaminated zone soil concentrations.

1. EMDF waste concentrations (C_{waste}; pCi/g) for the initial inventory of all radionuclides at the assumed closure date (2047) were used (PA Appendix I, Table I.1). Per text of the last paragraph of Section I.2, these concentrations were then multiplied by 0.531 to account for fill used during emplacement of the waste, yielding average radionuclide concentrations in the EMDF disposal cell (C_{EMDF}). (Note that radionuclide concentrations in Table I.1 do not account for radiological ingrowth, and therefore underestimate initial concentrations of shorter-lived radionuclides in radionuclide decay chains.)

$$C_{EMDF} = C_{waste} \times 0.531$$

2. Concentrations in garden soil (C_{garden}), based on average radionuclide concentrations in the EMDF disposal cell (, were calculated according to the method described in Section I.3 of Appendix I (tilling and dilution of contaminated drill cuttings into garden soil). A tilling depth of 0.15 m, rather than the value of 0.305 m specified in Section I.3, was used for consistency with the Base Case exposure model and the default soil mixing layer

assumption in RESRAD-ONSITE and RESRAD-OFFSITE. RESRAD-ONSITE contaminated zone (C_{garden}) concentrations were calculated as:

$$C_{garden} = (C_{EMDF} \times V_{borehole}) / (Depth_{till} \times Area_{garden})$$

where

$V_{borehole} = 1.96\ m^3$ (Appendix I, Section I.3)
$Depth_{till} = 0.15$ m (Appendix G, Table G.13)
$Area_{garden} = 2023\ m^2$ (Appendix I, Section I.3: 0.5 acre)

Several other RESRAD-ONSITE inputs were necessarily modified to create the input file for the RESRAD-ONSITE Chronic Post-Drilling Resident scenario for exposure to drill cuttings. The parameterization of inputs to the infiltration rate were not altered from the water pathways values, so the CZ (garden soil) is leached of soluble radionuclides over time in the same manner as the disposal cell in the water-pathways model. However, no water pathways are activated in the drill cuttings model, since these results are intended to be additive to those of the RESRAD-ONSITE water pathways model.

The need to run two separate RESRAD models to evaluate the Onsite Chronic Post-Drilling Resident scenario reveals a limitation of the RESRAD code. Pathways related to drill cuttings cannot be integrated into a single system model, because the code does not support direct specification of a drilling scenario. A probabilistic analysis of the coupled model with both water and drill cuttings pathways is therefore impossible.

Inputs to the Onsite Chronic Post-Drilling Resident water pathways model (in addition to the CZ soil concentrations) that were changed to create the Onsite Chronic Post-Drilling Resident drill cuttings model include:

- pathways: external gamma, inhalation, plant ingestion, meat ingestion, soil ingestion
- Area of CZ: 2,023 m^2
- Thickness of CZ: 0.15 m
- Length parallel to aquifer: 50 m (square root of Area of CZ)
- Cover depth: 0 m
- Contaminated zone erosion rate: 0 m/year
- Livestock water: 0 m
- Irrigation water: 0 m

B6.0 References

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A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee:

Appendix C
Mercury Transport
RESRAD-OFFSITE Simulations

1. Title: A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee: Appendix C Mercury Transport RESRAD-OFFSITE Simulations		
2. Filename: EMDF PA Review Appendix C_FINAL.docx		
3. Description:		
	Name	Date
4. Originator	Ralph Perona	21 Nov 2018
5. Reviewer	Terry Jennings, Paul Black, and Chris Schaupp	3 Dec 2018
6. Remarks		

CONTENTS

FIGURES

C1.0 Introduction

Neptune performed simulations to evaluate the characteristics of mercury transport using the RESRAD-OFFSITE code that was employed to assess radiological transport and dose in the EMDF PA. The issue of the exclusion of non-radiological waste constituents in the EMDF PA modeling was described in Critical Issue 7 of the Neptune and Company, Inc. (Neptune) review of the EMDF PA and CA. This appendix provides technical details in support of the summary of Key Findings pertaining to Critical Issue 7 in Section 2.2 of Neptune's EMDF PA/CA review. Mercury was selected for this evaluation because it is anticipated to be the most significant non-radiological constituent in terms of mass and potential risks in the waste stream identified for disposal at the EMDF.

Mercury was evaluated by using the very long-lived radioisotope Zr-93 as a surrogate. Analogous to the EMDF PA base case modeling, these simulations used the Version 2 release model, with the leach rate calculated internally based on the solid-water partition coefficient (K_d). A range of long-term infiltration rates (the PA base case as-built estimate of 0.43 inch/year, the long-term PA degraded rate base case of 1 inch/year, and also an infiltration rate of 2 inches/year, which is one-quarter of the native recharge rate of approximately 8 inches/year) was used to evaluate sensitivity of mercury leaching and transport to this important assumption. Additionally, the RESRAD default mercury K_d of 52 cm^3/g was replaced with a best-estimate value from independent review of the literature (11.6 cm^3/g) to gauge sensitivity of mercury transport to uncertainty in the mercury K_d. The RESRAD-OFFSITE model developed in this review was run out to 25,000 years to gain an understanding of very-long-term system performance.

To evaluate the sensitivity of mercury transport to uncertainty in the mercury K_d, a second value was obtained from a reference other than RESRAD. A mercury K_d of 11.6 cm^3/g was taken from a database of K_d values developed by Neptune for supporting probabilistic PA modeling. The value is the geometric mean of 35 records for the mercury K_d. All but one of these 35 records represent a K_d value that was used by a subject matter expert in a specific application, and the remaining record is an experimental value. This additional mercury K_d value was used to gauge the influence of uncertainty in mercury K_d on mercury transport.

The radionuclide Zr-93 was selected to represent the environmental transport of mercury. The half-life of Zr-93 (1.5E+06 y) provides for essentially zero decay over the 10,000-year EMDF PA modeling period, which is necessary to emulate a stable element. The Zr K_d used in the EMDF (50 cm^3/g) was changed to the default RESRAD mercury K_d (52 cm^3/g), and a second K_d value of 11.6 cm^3/g was also used as discussed above. The default RESRAD-OFFSITE transfer factors for mercury were also applied to use Zr as a surrogate for mercury. A screen shot of the element-specific RESRAD-OFFSITE default parameter values for mercury that were applied to Zr-93 in these simulations are shown in Figure C 1 below:

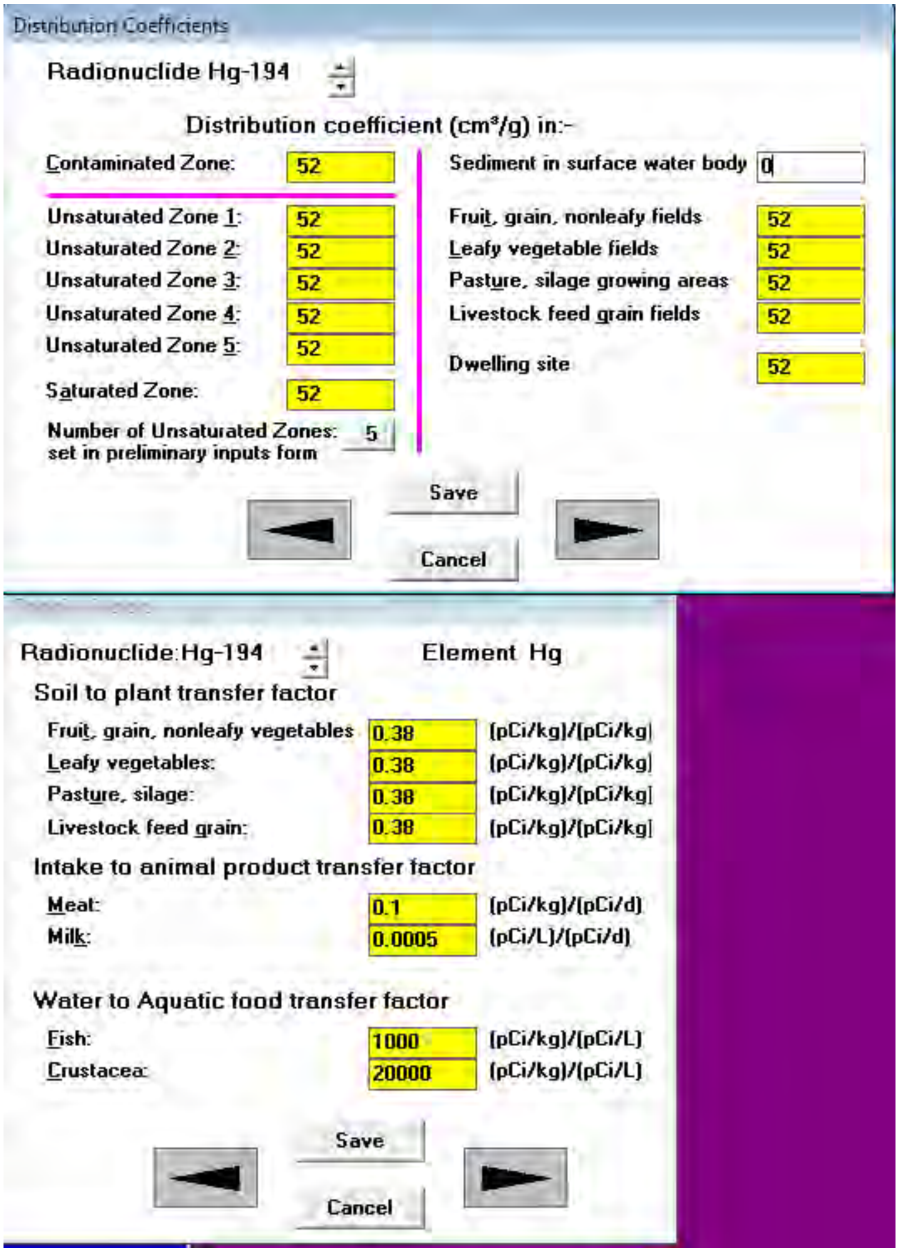

Figure C 1. Screen capture of RESRAD-OFFSITE mercury transport parameter values.

An initial source concentration of 15 pCi/g mercury (Zr-93) was used in the RESRAD-OFFSITE evaluation of environmental mercury transport. This initial bulk waste concentration was determined as follows:

The dimensions of the "contaminated zone" of the EMDF landfill that are used in the EMDF PA RESRAD modeling are 92,590 m^2 and 18.288 m thick. Translating to cubic yards, at 1.308 yd^3 per m^3, returns a volume of 2.2E+06 yd^3. Table 2-5 of the CERCLA RI/FS (DOE/OR/01-2535&D5, 2/8/2017) provides as-disposed estimates of 60,553 yd^3 for Hg-contaminated debris, and 42,047 yd^3 for Hg-contaminated soil. The table also assumes a 25% uncertainty in total as-disposed volume (not applied to individual waste streams). Applying this uncertainty to these estimated mercury-contaminated waste stream volumes results in a protective estimate of 128,250 yd^3. Therefore, an estimated fraction of mercury-containing waste in the entire volume of the closed landfill is 128,250 / 2.2E+06, or 5.8%.

40 CFR Section 268.40 (Part 268-Land Disposal Restrictions, Subpart D-Treatment Standards, Applicability of Treatment Standards) states:

> … for mercury, the 0.025 mg/l TCLP standard that meets 268.48 standards applies to: all other nonwastewaters that exhibit, or are expected to exhibit, the characteristic of toxicity for mercury based on the toxicity characteristic leaching procedure (TCLP) in SW846; and contain less than 260 mg/kg total mercury.

Averaging the mercury-containing waste within the entire landfill, a bounding contaminated zone source term for RESRAD transport evaluation of 260 mg/kg × 0.058, or 15 mg/kg mercury, is derived.

Ultimately, it's desirable to compare RESRAD-OFFSITE well and surface water concentrations to the 51 ng/L mercury standard for Bear Creek. Note that 15 mg/kg = 0.015 mg/g, which in turn = 15,000 ng/g. Substituting pCi for ng, a value of 15,000 pCi/g of Zr-93 (mercury) was applied as the Contaminated Zone soil concentration in RESRAD-OFFSITE. RESRAD output for concentrations in well water and groundwater (in pCi/L) can then be directly interpreted as nCi/L.

All radionuclides other than C-14, I-129, Tc-99, and Zr-93 were removed for the Mercury Transport simulations in order to improve computational efficiency—a deterministic model run with these four radionuclides was executed in 20 seconds, versus a run time of approximately 17 minutes for the PA Base Case with all radionuclides. The original concentrations of C-14, I-129, and Tc-99 (which contributed ~100% of Base Case doses) were included in order to verify that the Mercury Concentrations simulation is identical to Base Case by comparison of the dose results for those radionuclides. Dose results over time for C-14, I-129, and Tc-99 were confirmed as identical to those in the Base Case simulation.

C2.0 Results

1. Using the RESRAD-OFFSITE default mercury K_d of 52 cm^3/g in the Base Case model, and the as-built infiltration rate of 0.43 in/year, breakthrough to groundwater for mercury occurred at approximately model year 8,100, and surface water concentrations reached only 1E-06 ng/L at model year 25,000. This concentration is far below the 51 ng/L mercury standard for Bear Creek.
2. Using the RESRAD-OFFSITE default mercury K_d of 52 cm^3/g in the Base Case model, and the long-term degraded rate of 1 in/year, breakthrough to groundwater for mercury occurred at approximately model year 4,600, and surface water concentrations reached the 15 ng/L mercury standard at model year 25,000. With an infiltration rate of 2 in/year, mercury surface water concentrations reach 1 ng/L about year 15,000 and 50 ng/L about year 19,000.

3. Using a mercury K_d of 11.6 cm^3/g dramatically affects the simulation results for surface water concentrations. With a K_d of 11.6 cm^3/g and 0.43 in/year of infiltration, surface water mercury concentrations reach 36 ng/L at year 10,000, hit 50 ng/L about year 10,350, and peak at about 190 ng/L around model year 15,000.
4. Using a mercury K_d of 11.6 cm^3/g and the long-term base case degraded state infiltration rate of 1 in/year provides a good evaluation of the sensitivity of mercury transport to its K_d value. With 1 in/year of infiltration, surface water mercury concentrations reach 50 ng/L at year 5,500, and peak at about 430 ng/L around model year 8,500. An accurate understanding of the K_d value for the form(s) of mercury to be disposed at the EMDF, and the associated uncertainties, will be critical for evaluating mercury disposal in a risk-based context. Naturally, using the higher 2 in/year assumption for long-term infiltration rate results in mercury surface water concentration peaks that are earlier and larger: the peak concentration of about 810 ng/L is reached around model year 6,200.

The results of the RESRAD-OFFSITE simulations for surface water mercury concentrations using different assumed values of infiltration rate and mercury K_d are shown in Figure C 2:

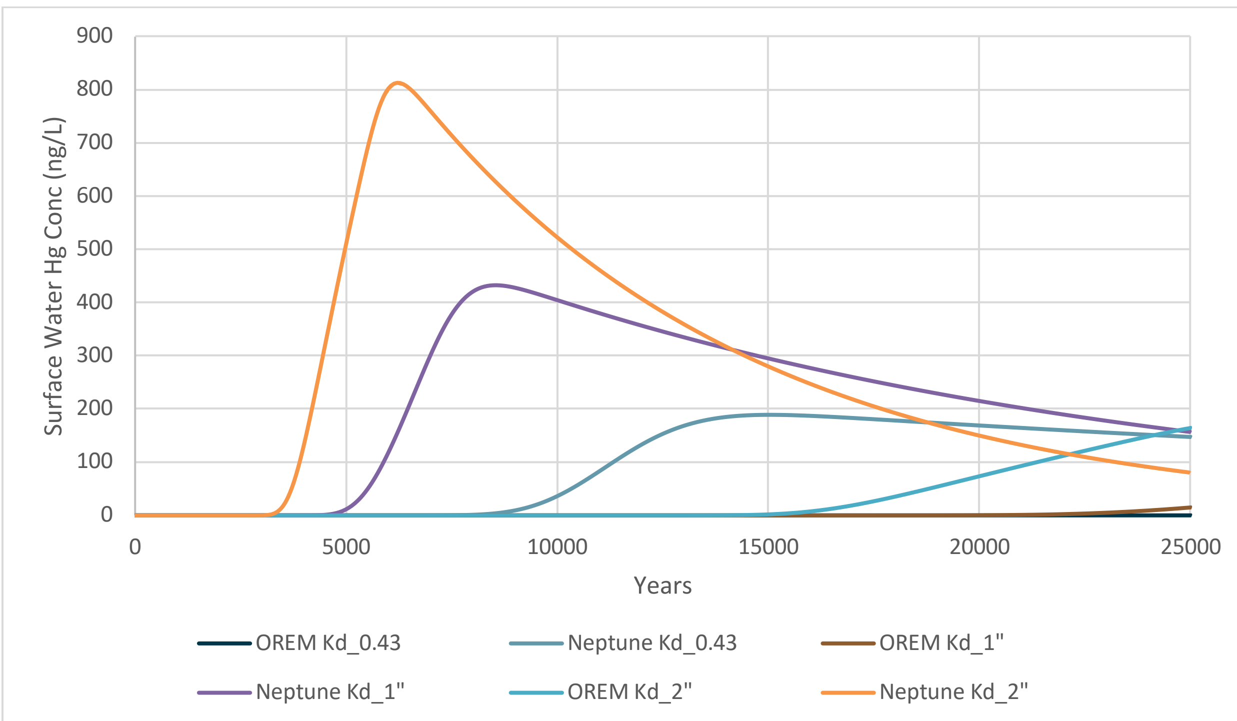

Figure C 2. Mercury surface water concentrations in Bear Creek modeled in RESRAD-OFFSITE with different infiltration rate and K_d values.

Note that neither the RESRAD nor the Neptune mercury K_d value may be applicable to mercury in the event that encapsulation or other treatment affects the partitioning behavior of mercury in the Y-12 waste. If elemental mercury is present in the waste these simulations are likely to be wholly inapplicable, as elemental mercury is insoluble in water but might migrate as a vapor or as a free-phase liquid at environmental temperatures. The chemical form of mercury released from the waste, especially whether it is elemental or as a chemical compound, will therefore greatly influence fate and transport.

With regard to the RESRAD-OFFSITE water pathways calculations, these uncertain inputs have a greater-than-linear influence on results: 1) the infiltration rate, and 2) the element-specific K_d value. A defensible estimation of these values and associated uncertainties is therefore critical to modeling of any constituent, including mercury. The selected point of compliance for surface water concentrations (for example, in a lateral drainage subsequent to bathtubbing or in Bear Creek), will also affect interpretation of these mercury transport simulations.

C2.1 Supporting Documentation of RESRAD Leach Rate and Infiltration Rate Calculation

This section of the appendix provides some supporting documentation for specification of RESRAD-OFFSITE inputs to return different infiltration rate values. As discussed above, simulations were conducted using infiltration rate values of 0.43 in/year, 1 in/year, and 2 in/year. Per Section 3.3.3.2 of the EMDF PA, "The initial and final leach rates have units of 1 /year and are calculated as the reciprocal of the retarded vertical travel time through the waste, based on values of cover infiltration, the moisture content, bulk density, and average thickness of the waste, and the assumed (base case) value of K_d for each radionuclide (refer to Sect. G.4.3.5 of PA Appendix G)."

The RESRAD-OFFSITE Version 2 first order leach rate equation is documented in *New Source Term Model for the RESRAD-OFFSITE Code Version 3* NUREG report (Yu et al. (2013), eq 2.42):

The user can specify one or both of the inputs that become active when this option is chosen; "Specify First Order Leach Rate" and "Use Distribution Coefficient to Estimate First Order Leach Rate." If the Specify First Order Leach Rate input is left unchanged at zero, the code will estimate that leach rate by using the expression:

$$\mu = \frac{1}{\theta_{pc} + K_d \rho_b} \frac{I}{T_{pc}(0)}. \tag{2.42}$$

where

μ = first-order leach rate (yr^{-1}),
I = infiltration rate ($m\ yr^{-1}$),
θ_{pc} = total moisture content of the primary contamination,
K_d = distribution coefficient ($cm^3\ g^{-1}$),
ρ_b = bulk density of the primary contamination ($g\ cm^{-3}$), and
$T_{pc}(0)$ = initial thickness of the primary contamination (m).

where I is calculated as documented in Section G.4.3.5.2 of the EMDF PA, which is Equation 3.18 of the RESRAD-OFFSITE Version 2 Manual (Yu et al. 2007):

3.2.1.12 Pore Water Velocity

The volumetric flow rate through a unit cross section is called the Darcy velocity or apparent velocity of flow. The Darcy velocity of flow in the partially saturated zone is the infiltration rate, which is computed by using the following expression:

$$V_d = I = (1 - C_e)\left[(1 - C_r)P_r + I_{rr}\right] , \qquad (3.18)$$

where

V_d = Darcy velocity (m year^{-1}),

I = infiltration rate (m year^{-1}),

C_e = evapotranspiration coefficient,

C_r = runoff coefficient,

P_r = precipitation rate (m year^{-1}), and

I_{rr} = annual irrigation applied over the primary contamination (m year^{-1}).

The EMDF PA references its leach rate calculation to the RESRAD-ONSITE user manual (Equation E-3 of Appendix E of Yu et al. (2001)), but this appears equivalent to Equation 2.42 of Yu et al (2013). From Yu et al. (2001):

The first-order leach rate constant used in the current version of RESRAD is a time-independent radionuclide leach rate constant that is estimated on the basis of the soil residence time for the initial thickness of the contaminated zone. A time-independent radionuclide leach rate constant for radionuclide i, L_i, may be written as

$$L_i = \frac{I}{\theta^{(cz)}\ T_0\ R_{d_i}^{(cz)}} , \qquad (E.3)$$

where

I = infiltration rate (m/yr),

$\theta^{(cz)}$ = volumetric water content of the contaminated zone,

T_0 = initial thickness of the contaminated zone (m), and

$R_{d_i}^{cz}$ = retardation factor in the contaminated zone for radionuclide i (dimensionless).

and,

The retardation factor for radionuclide i, R_{d_i}, is the ratio of the average pore water velocity to the radionuclide transport velocity. On the basis of the assumption that the adsorption-desorption process can be represented with a linear isotherm, the retardation factor can be calculated with the following formula (Yu 1987):

$$R_{d_i} = 1 + \frac{\rho_b K_{d_i}}{\theta} = 1 + \frac{\rho_b K_{d_i}}{p_t R_s} , \qquad \text{(E.8)}$$

In these simulations, the Version 2 RESRAD option of calculating the first-order leach rate using the distribution coefficient was selected. The infiltration rates in RESRAD were calculated using Equation 3.18 of the RESRAD-OFFSITE manual (equivalent to Equation E.4 of the RESRAD-ONSITE manual). More specifically, values of the evapotranspiration and runoff coefficients were selected to return the range of infiltration rates used in these simulations, as shown in a screen shot from the Excel workbook used for the calculations:

from the RESRAD 6 Manual, Eq E.4: $I = (1 - C_e)[(1 - C_r)P_r + I_{irr}]$

where:

C_e = evapotranspiration coefficient

C_r = runoff coefficient

P_r = precipitation rate

I_{irr} = irrigation rate

	0.43"	1"	2"	4"
C_e	0.815	0.568	0.6	0.6
C_r	0.957	0.957	0.908	0.815
P_r	1.382	1.382	1.382	1.382
I_irr	0	0	0	0
calculated:	0.43	1.0	2.0	4.0
inch / m	39.37			

C3.0 References

Yu, C., et al., 2007. *User's Manual for RESRAD-OFFSITE Version 2* ANL/EVS/TM/07-1, DOE/HS-0005, NUREG/CR-6937, prepared for the United States Department of Energy and the United States Nuclear Regulatory Commission, Argonne National Laboratory, Argonne IL, June 2007

Yu, C., et al., 2013. *New Source Term Model for the RESRAD-OFFSITE Code Version 3,* NUREG/CR-7127, ANL/EVS/TM/11-5, prepared by Argonne National Laboratory, United States Nuclear Regulatory Commission, Washington DC, December 2013

Yu, C., et al., 2001. *User's Manual for RESRAD Version 6,* ANL/EAD-4, prepared for the United States Department of Energy, Argonne National Laboratory, Argonne IL, July 2001

A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee:

Appendix D
Radon Flux Calculations—Review and Supplemental Evaluation

1. Title: A Review of the Draft Performance Assessment and Composite Analysis for the Proposed Environmental Management Disposal Facility, Oak Ridge, Tennessee: Appendix D Radon Flux Calculations—Review and Supplemental Evaluation		
2. Filename: EMDF PA Review Appendix D_FINAL.docx		
3. Description:		
	Name	Date
4. Originator	Ralph Perona	21 Nov 2018
5. Reviewer	Terry Jennings, Paul Black, and Chris Schaupp	4 Dec 2018
6. Remarks		

CONTENTS

TABLES

D1.0 Introduction

Radon flux calculations for the EMDF PA are described in Appendix H of the EMDF PA. These calculations were performed using the equations described in *Radon Attenuation Handbook for Uranium Mill Tailings Cover Design* (Rogers et al. 1984) and implemented in Microsoft Excel. As part of its review of the EMDF PA, Neptune performed independent Radon flux calculations using the RESRAD-ONSITE (version 7.2) computer code. The purpose of these independent calculations is threefold:

1. to benchmark the NRC radon flux calculations using a different methodology,
2. to investigate radon flux under the bounding conditions of cover homogenization (functional loss of the integrity of the cover clay liner), and with consideration for cover loss by erosion, and
3. to evaluate potential radon dose under the chronic inadvertent human intruder (IHI) scenario, which assumes a residence located on the EMDF.

Neptune's independent evaluation of radon transport is described in detail in Section 2.3 of Neptune's review of the EMDF PA and CA. This appendix provides technical details in support of the summary in Neptune's PA and CA review.

D2.0 Key Findings

1. Employing the EMDF PA base case assumption that combined amended and cover compacted clay layers maintain a water saturation of 99% in perpetuity, radon flux calculations using RESRAD-ONSITE confirm the EMDF PA conclusion of approximately zero radon flux through this layer. Using lower water saturation levels of 90% and 75%, with all other EMDF PA radon modeling assumptions unchanged, calculated radon flux at model year 1,000 was 6.5E-04 pCi/m^2-s and 0.13 pCi/m^2-s, respectively. Even at model year 10,000, as the immediate parent of radon (radium-226) ingrows from uranium-234, radon flux was below the 20 pCi/m^2-s regulatory standard with moisture contents of 90% (0.013 pCi/m^2-s) and 75% (2.7 pCi/m^2-s). This analysis indicates that the EMDF PA conclusion that radon surface flux will remain below the 20 pCi/m^2-s regulatory standard is robust to different assumed water saturation levels for the compacted clay layer.

2. The impact of more severe cover degradation was evaluated by considering the condition of a homogenized cover, with uniform properties over a 10-ft thickness calculated as the weighted average of the properties (moisture content, density, porosity) of the individual layers. Radon flux through a 10-ft homogenous cover was calculated for a zero-erosion condition (the EMDF PA base case assumption) and with an erosion rate selected from the EMDF PA erosion sensitivity analysis (PA Appendix C, Table C.7). Radon flux through the 10-ft homogenized cover remains below the 20 pCi/m^2-s standard through 10,000 years if zero cover erosion is assumed. Using a long-term cover erosion rate of 0.0732 cm/year, the 10-ft cover is removed after approximately 4,100 years but radon flux does not exceed 20 pCi/m^2-s until between 5,000 and 10,000 years, where it reaches 35 pCi/m^2-s at 10,000 years. This analysis indicates that ingrowth of radium-226 over a time period of more than

5,000 years is necessary in order to reach the 20 pCi/m^2-s regulatory standard even if no cover material is present above the waste.

3. The radon dose for a residential (IHI) receptor in a slab-on-grade home on top of the EMDF was calculated with RESRAD-ONSITE assuming a homogenized 10-ft cover under both zero-erosion and 0.0732 cm/year erosion assumptions. These calculations indicate a radon dose at year 1,000 of approximately 1 mrem/year (zero erosion) and 2.6 mrem/year (0.0732 cm/year erosion). Residential radon dose assuming slab-on-grade construction remained below the IHI 100 mrem/year threshold for 10,000 years. In the erosion scenario, radon dose reaches the 100 mrem/year threshold after model year 3,000 and is approximately 1,200 mrem/year at 10,000 years.

D3.0 Details of the Supplemental Evaluation

RESRAD-ONSITE was run with only the Radon pathway activated. Initial assumed contaminated zone concentrations (Activity at disposal) of U-238 [182 pCi/g], U-234 [628 pCi/g], Th-230 [1.66 pCi/g] and Ra-226 [0.712 pCi/g] were taken from PA Appendix H, Table H.3 and confirmed by comparison to PA Appendix G, Table G.9.

RESRAD-ONSITE was used to evaluate the radon flux through the clay barrier and compared to results described in Appendix H of the EMDF PA. Lower values for clay water saturation content than those assumed in Appendix H (99% for the clay layer group) were also used to investigate sensitivity to this assumption. In principle, at levels of very high-water saturation, the diffusion of radon in the water phase could be an important contributor to flux through the clay barrier, but this has not been evaluated. Conceptually, these lower saturations could be representative of future conditions where the integrity of the barrier has been compromised in some manner. Inputs to the clay barrier radon flux calculations are shown below in Table D 1, and a comparison of RESRAD-OFFSITE flux estimates to that calculated in Appendix H of the EMDF PA is provided below in Table D 2.

Table D 1. RESRAD-ONSITE Radon Pathway Parameter Values—Clay Barrier Flux (“Outdoor” Flux).

Parameter	Units	RESRAD Default	EMDF Value	Reference / Note
Cover / Hydrology Inputs				
Thickness of contaminated zone	m	2	15	Table H.2; waste layer
Cover depth (clay barrier)	m	–	0.61	Table H.2; amended and cover clay layers, 1-ft thickness of each
Density of cover material	g/cm2	1.5	1.52	Table H.2; amended and cover clay layers
Cover erosion rate	m/year	0.001	0	Base case assumption
Density of contaminated material	g/cm2	1.5	1.54	Table H.2; waste layer

Parameter	Units	RESRAD Default	EMDF Value	Reference / Note
Contaminated zone erosion rate	m/year	–	0	Cover exists throughout modeling period
Contaminated zone total porosity	–	0.4	0.419	Table H.2; waste layer
Evapotranspiration coefficient	–	0.5	0.999	Maximum allowed value, defines a static contaminated zone with no radionuclide loss from leaching
Irrigation	m/year	0.2	0	
Radon Pathway Inputs				
Cover (clay) total porosity	–		0.427	Table H.2; amended and cover clay layers
Cover (clay) volumetric water content	–		0.423; 0.383; 0.320	Test runs at 99%, 90%, and 75% saturation
Cover (clay) radon diffusion coefficient	m2/s		-1	RESRAD calculation based on porosity and water content
Contaminated zone (waste) radon diffusion coefficient	m2/s		-1	RESRAD calculation based on porosity and water content
Rn-222 radon emanation coefficient	–	0.25	0.25	The EMDF PA (Section H.6) references RESRAD for this value

Table D 2. Comparison of RESRAD-ONSITE and PA Appendix H Radon Flux Through the 2-ft Thick Compacted Clay Barrier.

	Radon Flux (pCi/m^2-s)			
Time	**EMDF PA, Table H.4 (99% saturation)**	**RESRAD-ONSITE; 99% saturation[1]**	**RESRAD-ONSITE; 90% saturation**	**RESRAD-ONSITE; 75% saturation**
100 years	–	0	2.3E-04	0.048
1,000 years	1.05E-06	0	6.5E-04	0.13
5,000 years	–	0	0.0055	1.1
10,000 years	–	0	0.013	2.7

[1]With an assumed 99% water saturation, even when replacing the RESRAD-calculated effective radon diffusion coefficient (1.28E-05 cm^2/s) with the value used in the EMDF PA (5.96E-05 cm^2/s) zero radon flux is returned in RESRAD-ONSITE.

RESRAD-ONSITE was also used to evaluate radon flux through a naturalized (homogenized) cover, where a single volumetric water content was estimated by weighting the volumetric water content of each layer according to the thickness of that layer within a 10-ft cover. Using the thicknesses and saturation fractions in Table H.2, a weighted water saturation of approximately 50% was calculated. A weighted total porosity of 0.43 and a weighted dry bulk density of 1.38 g/cm^2 were similarly calculated. Practically, this homogenization is intended to represent conditions where the intactness of the clay barrier has been compromised by one or more processes, such as described in PA Appendix C, Section C.1.2.2.2: "In the post-closure period, differential settlement of the waste and overlying cover components can impair performance in the absence of corrective maintenance. Eventually, severe weather events and progressive climate and vegetation changes can lead to erosion of the protective cover components and accelerate degradation of the clay barrier in the cover, increasing the likelihood of greater water infiltration over time." Inputs to the 10-ft homogenized cover radon flux calculations are shown below in Table D 3, and a comparison of these RESRAD-OFFSITE flux estimates to the EMDF PA clay barrier flux is provided in Table D 4.

Table D 3. RESRAD-ONSITE Radon Pathway Parameter Values—Homogenized Cover Flux ("Outdoor" Flux).

Parameter	Units	RESRAD Default	EMDF Value	Reference / Note
Cover / Hydrology Inputs				
Thickness of contaminated zone	m	2	15	Table H.2; waste layer
Cover depth	m	–	3.048	Table H.2
Density of cover material	g/cm2	1.5	1.51	Table H.2; weighted
Cover erosion rate	m/year	0.001	0	Base case assumption
Density of contaminated material	g/cm2	1.5	1.54	Table H.2; waste layer
Contaminated zone erosion rate	m/year	–	0	Cover exists throughout modeling period
Contaminated zone total porosity	–	0.43	0.419	Table H.2; waste layer
Evapotranspiration coefficient	–	0.5	0.999	Maximum allowed value, defines a static contaminated zone with no radionuclide loss from leaching
Irrigation	m/year	0.2	0	
Radon Pathway Inputs				
Cover total porosity	–		0.43	Table H.2; weighted
Cover volumetric water content	–		0.215	Table H.2; weighted
Cover (clay) radon diffusion coefficient	m2/s		-1	RESRAD calculation based on porosity and water content
Contaminated zone (waste) radon diffusion coefficient	m2/s		-1	RESRAD calculation based on porosity and water content
Rn-222 radon emanation coefficient	–	0.25	0.25	The EMDF PA (Section H.6) references RESRAD for this value

Table D 4. Comparison of RESRAD-ONSITE Clay Barrier Radon Flux and PA Appendix H Radon Flux Through a Homogenized Cover with 50% Water Saturation.

	Radon Flux (pCi/m²-s)		
Time	**EMDF PA, Table H.4**	**RESRAD-ONSITE; 10-ft (3-m) cover, no erosion**	**RESRAD-ONSITE; 10-ft (3-m) cover, 0.0732 cm/year erosion [a]**
100 years	–	0.0082	0.0090
1,000 years	1.05E-06	0.023	0.060
5,000 years	–	0.19	11 [b]
10,000 years	–	0.47	35 [b]

[a] Table C.7, RUSLE2 sensitivity analysis, Alfalfa hay (harvested). Used to represent unknown long-term side slope erosion of unmaintained forest with periodic fires.
[b] Side slope cover is completed eroded after 4,100 yr.

Radon doses for a receptor on top of the EMDF were calculated using RESRAD-ONSITE for the homogenized cover conditions described in Table D 4. As shown in Table D 2, RESRAD-ONSITE estimates zero radon flux through the compacted clay barrier using the base case assumption of 99% water saturation. The default occupancy parameters (30-yr exposure duration, 50% indoor time fraction, 25% outdoor time fraction) and residential building characteristics in the RESRAD-ONSITE computer code, version 7.2, were used in these calculations.

Resident radon doses over time for a home situated on the EMDF are shown below in Table D 5. These doses are applicable in principle to an IHI “chronic post-drilling” scenario, such as that described in Appendix I of the EMDF PA. Although the scenario defined in Appendix I is limited to soil and food exposures subsequent to mixing of drill cuttings in garden soil, the presence of drill cuttings and a garden implies the existence of a home and other potentially complete exposure pathways, including indoor inhalation of radon.

Table D 5. Radon Dose for a Resident on a Homogenized Cover with 50% Water Saturation.

	Radon Dose (mrem/year); slab-on-grade construction [a]	
Time	**RESRAD-ONSITE; 10-ft (3-m) cover, no erosion**	**RESRAD-ONSITE; 10-ft (3-m) cover, 0.0732 cm/year erosion [b]**
100 years	0.35	0.39
1,000 years	0.98	2.6
2,000 years	2.3	16
3,000 years	4.1	76
4,000 years	6.2	300
5,000 years	8.4	490 [c]
10,000 years	20	1,200 [c]

[a] A higher dose would be received in a home with basement.
[b] Table C.7, RUSLE2 sensitivity analysis, Alfalfa hay (harvested). Used to represent unknown long-term erosion of unmaintained forest with periodic fires.
[c] Cover is completed eroded after 4,100 yr.

D4.0 References

Rogers, V.C., et al., 1984. *Radon Attenuation Handbook for Uranium Mill Tailings Cover Design,* NUREG/CR-3533, PNL-4878, RAE-18-5, United States Nuclear Regulatory Commission, Washington DC, April 1984

Enclosure 2

TDEC Review (June 15, 2020)
Draft Waste Acceptance Criteria
Proposed Environmental Management Disposal Facility
Oak Ridge, Tennessee

The Tennessee Department of Environment and Conservation (TDEC) offers this review to support dialog with the U.S. Department of Energy (DOE) - Oak Ridge Office of Environmental Management (OREM). The goal is to develop protective Waste Acceptance Criteria (WAC) for the proposed Environmental Management Disposal Facility (EMDF) before OREM issues a draft (D1) Record of Decision (ROD). Protective WAC are needed to show that EMDF would protect people from radiation-induced cancer and health effects from hazardous & toxic chemicals.

In 2018, OREM issued a proposed plan[1] that describes TDEC's concerns, including the need to evaluate WAC to protect people from exposure to radioactive material, now and in the future. In 2019, OREM provided draft WAC information from the forthcoming ROD and requested TDEC's feedback.[2] TDEC engaged Neptune and Company, Inc. (Neptune) as subject matter experts (SMEs) to review OREM's proposed WAC and determine whether the criteria meet the requirements of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).[3] Neptune has extensive experience with complex radioactive waste-disposal challenges faced by DOE and other entities throughout the U.S. and around the world. Neptune identified opportunities for improving the proposed WAC.

The following pages summarize the findings of the preliminary review. TDEC may offer additional feedback following review of the recently finalized Performance Assessment (PA)[4] and Composite Analysis (CA)[5]. Those reports should provide information vital to understanding the protectiveness of the proposed WAC.

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[1] *Proposed Plan for the Disposal of Oak Ridge Reservation Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) Waste* (DOE/OR/01-2695&D2/R1); available at https://doeic.science.energy.gov/uploads/A.0100.030.2596.pdf.

[2] Draft WAC information from the forthcoming *Record of Decision for Comprehensive Environmental Response, Compensation, and Liability Act Oak Ridge Reservation Waste Disposal at the Environmental Management Disposal Facility, Oak Ridge, Tennessee* (DOE/OR/01-2794&D1) (shared with TDEC on July 30, 2019).

[3] Commonly known as "Superfund," CERCLA is the Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986.

[4] *Performance Assessment for the Environmental Management Disposal Facility at the Y-12 National Security Complex, Oak Ridge, Tennessee* (UCOR-5094/R2); April 28, 2020 (shared with TDEC on May 18, 2020).

[5] *Composite Analysis for the Environmental Management Waste Management Facility and the Environmental Management Disposal Facility, Oak Ridge, Tennessee* (UCOR-5095/R2); April 29, 2020 (shared with TDEC on May 18, 2020).

TN Department of Environment & Conservation

Administrative WAC

- Draft ROD text does not describe how DOE would manage the waste to avoid spontaneous criticality, a self-sustaining nuclear chain reaction.

- Draft ROD text does not say whether EMDF would accept hazardous waste, nor does it discuss waste treatment to stabilize and/or immobilize hazardous chemicals like mercury. Missing information includes contaminants that may require treatment, anticipated waste forms, and proposed limits on hazardous contaminants. All are critical to understanding how EMDF would protect people and the environment, now and in the future.

- The ROD must state that mixed waste disposed of in the proposed EMDF will be treated using technology approved by TDEC under the existing *Site Treatment Plan for Mixed Wastes on the U.S. Department of Energy Oak Ridge Reservation* (STP).

- The proposed WAC mentions "*free-flowing liquids*". This phrase is not defined in regulations, although the term "free liquid" is defined and linked to a specified test method in RCRA Subtitle C regulations applied as ARARs. Source separation should be employed to recover elemental mercury as a liquid. There are alternate treatment methods for debris, soils and high- and low-concentration categories of RCRA D009 mercury characteristically hazardous waste. Compliance with ARARs related to land disposal restrictions (LDRs) for mercury and the regulations prohibiting disposal of liquids in a hazardous waste should be better delineated and described for different forms of mercury waste and contaminated media.

- Limits on polychlorinated biphenyl (PCB) liquid disposal need clarification to ensure compliance with TSCA ARARs. As currently worded, it is not clear that liquids with more than 50 parts per million (ppm) PCBs must be treated so they will not flow and that liquids with more than 500 ppm PCBs must not be placed in the landfill under any circumstances.

Analytic WAC

- Draft ROD text does not include analytic WAC for hazardous and toxic chemicals likely to be in EMDF waste, such as mercury & PCBs. WAC may be necessary to decrease concentrations of such chemicals in landfill wastewater to meet ROD requirements for treating landfill wastewater to protective levels before discharging it into local streams. DOE should evaluate whether WAC limits are needed for chemicals like mercury to ensure selected technologies can treat landfill wastewater to protective discharge levels.

- It is not clear the draft WAC account for radionuclides that contribute substantially to the radiation dose the public could receive from the waste or from wastewater that flows downstream. Some progeny (nuclides produced through radioactive decay) produce more radiation dose than their parent radionuclides.[6] For such radionuclides, protecting the public requires more than limiting the amount of radioactivity the waste produces upon disposal. In such cases, protectiveness requires limits on the amounts (inventories) of specific radionuclides placed in the landfill. This is necessary to account for the radiation dose produced by the parents and their progeny over time *in addition to* the dose produced upon waste disposal.

  For example, determining safe amounts of plutonium-241 (Pu-241), americium-241 (Am-241), and neptunium-237 (Np-237) for disposal must account for the fact that Pu-241 (14-year half-life) decays to Am-241, which increases the amount of Am-241 in the waste. Americium-241 (half-life about 430 years) decays to Np-237, increasing the amount of that radionuclide. Neptunium-237 produces a lot more dose than Pu-241 or Am-241, so it poses a much greater threat. Moreover, Np-237 is a threat for a much longer time because it has a half-life greater than 2 million years. Failure to consider the amount of a specific radionuclide when setting limits for others in the same "decay chain" could result in unacceptable radiation doses to the public. It is not enough to set limits for each radionuclide in isolation; limits for all radionuclides in the decay chain must be considered based on the combined threat. If Am-241 were disposed at its WAC limit, that limit would be exceeded within a few decades as more Pu-241 decays and produces more Am-241. The same is true for Np-237, with an even longer half-life. There are many other examples of this this fundamental concept.

- Draft WAC include radioactivity limits for 48 radionuclides to ensure protectiveness if someone digs into the waste in future.[7] However, the text says when setting WAC to ensure protectiveness if there is a release from the landfill, OREM will track and limit the total amount (inventory) of only three radionuclides: tritium (H-3), technetium-99 (Tc-99), and carbon-14 (C-14) because PA model results show they are the only ones that may escape from the landfill and reach the highest concentrations within 1,000 years. This finding is unusual. PA models for most radioactive waste disposal facilities find that

[6] A nuclide is a distinct kind of atom. Some are stable, and others are radionuclides, meaning they are radioactive. Radionuclides have excess nuclear energy. They are unstable and decay to produce other nuclides, some of which may also be radioactive. In this case, dose refers to the radiation absorbed by people exposed to radionuclides in radioactive material.

[7] Table 2.6 lists 48 radionuclides, but the text says OREM considered 49 radionuclides in the assessment.

many more radionuclides may escape, even though the facilities are typically in drier (desert) locations in the western U.S., where lesser amounts of rain cannot flush contaminants into surrounding areas as easily as higher rainfall amounts can in Oak Ridge. The WAC should be modified to:

- **Track all radionuclides in the inventory.** The plan to track only three radionuclides is inadequate. OREM must track the amount of all radionuclides identified in waste profiles and placed in the proposed EMDF landfill to document the total inventory.[8]

- **Limit many radionuclides based on mobility, bioaccumulation or other risk factors.** OREM should limit the inventory of radionuclides with the greatest potential to harm streams, fish, and people because of their ability to travel, accumulate, or cause cancer or other health effects in the future. Attachment 1 presents examples of radionuclides that may need inventory limits to protect people who eat fish caught downstream of EMDF after DOE no longer controls the area.

- **Establish flexibility to limit additional radionuclides during operations based on new information.** There should be a means to add WAC requirements, including inventory limits, for additional radionuclides if needed. For example, radionuclides projected to have negligible inventories may be disposed of in greater quantities than originally anticipated, and additional radionuclides may be found to be significant at Oak Ridge National Laboratory (ORNL) or other waste sources.

- Draft ROD text says, "*Class C concentration limits are more restrictive (lower) than limits based on the EMDF intrusion performance analysis for 13 radionuclides (Table 2.6).*"[9] This suggests DOE's calculations would allow radioactive waste in the proposed EMDF to exceed Class C limits. The ROD must specify that OREM would not place Greater-Than-Class-C (GTCC) waste in the proposed EMDF. GTCC waste generally requires more protective disposal methods, such as burial in a deep geologic repository like the Waste Isolation Pilot Plant (WIPP) in New Mexico, to make it safe for public health and the environment.

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[8] OREM reports not knowing the true Environmental Management Waste Management Facility (EMWMF) inventory because it tracks only a few radionuclides for that landfill.

[9] The Nuclear Regulatory Commission (NRC) considers three classes of low-level radioactive waste suitable for burial in a landfill. Class A has the least radioactivity, most of which decays to background levels within a few decades. Class B has more radioactivity than Class A, and Class C has even more radioactivity than Class B. https://www.energy.gov/sites/prod/files/2019/06/f63/Nuclear-Regulatory-Commission%E2%80%99s-Low-Level-Radioactive-Waste-Classifications-June-2019.pdf

- Draft ROD text also suggests EMDF might allow disposal of waste that does not comply with WAC. OREM must not dispose of waste with radioactivity above compliance standards.

- The proposed WAC do not account for uncertainties in the projected amounts of radioactivity the waste would contain. OREM should follow common practice to manage uncertainty by using defensible statistical methods, such as the use of 95% confidence intervals instead of simply averaging projected levels of radioactivity. Establishing protective WAC requires DOE to understand how much of the waste may have above-average radioactivity levels.

- The proposed WAC are based on scenarios that do not assess increased public health risk from uranium's toxicity, which poses more potential short-term risk of health effects than uranium's cancer-causing radioactivity.

- The proposed WAC are also based on scenarios that do not assess public exposure to higher radioactivity and greater cancer risk because some radionuclides bioaccumulate in fish. Bear Creek sustains populations of rock bass and perch, both of which are fished for consumption.

- Draft ROD text proposes WAC without consideration of important natural processes like erosion. This is a significant deficiency because erosion could expose people to substantial radiation doses in the future by uncovering waste and carrying contamination downstream. The ROD should develop WAC based on evaluation of realistic erosion rates. Public exposure to eroded wastes would not depend on cap thickness or require someone to dig a residential basement into the waste, as suggested in the draft ROD text.

- A table in the draft ROD text presents limits on the amount of radioactivity someone could encounter upon accidentally discovering the waste in the future. However, the text does not explain how DOE established these limits. TDEC and the public need the DOE to "show your work" to verify the accuracy and protectiveness of these "intrusion-based" WAC.

<u>Operations-Based WAC</u>

- Along with DOE, TDEC & the U.S. Environmental Protection Agency (EPA) should be involved in defining and approving operations-based WAC. Consensus and transparency would eliminate conflicts of interest that exist when the party generating the waste has sole responsibility for deciding whether that material meets WAC at a landfill operated by the same party. Regulator involvement in the process, consistent with the Federal Facility Agreement for the Oak Ridge Reservation (FFA), would support the oversight needed to ensure WAC compliance.

Conclusion

TDEC is required by law to determine whether the ROD for the proposed landfill is meets CERCLA threshold criteria of protectiveness and compliance with ARARs. Before TDEC can approve a ROD that authorizes landfill construction, OREM must correct deficiencies in the WAC described above to show the EMDF would protect people from radiation-induced cancer, as well as health effects from hazardous & toxic chemicals.

Recommendations

In the spirit of partnership, the TDEC offers several recommendations for establishing protective WAC in the ROD and ensuring future waste disposal complies with those WAC.

- Obtain a Preliminary Disposal Authorization Statement (PDAS) demonstrating that DOE Headquarters finds the proposed EMDF would perform in a manner that protects the public from exposure to unacceptable radiation doses in accordance with DOE Orders. Before signing the ROD, TDEC needs to verify that PA & CA modeling used to develop WAC supports the requirements of CERCLA, including protecting people who catch and eat fish downstream.

- State in the ROD that the hazardous component of mixed waste disposed of in the proposed EMDF must be treated using technology approved by TDEC under the existing STP.

- Establish protective limits on the release of hazardous contaminants like mercury to local streams and/or treatment of waste before disposal.

- Protect receptors from uranium's non-cancer health effects, which pose more risk in the short term than its cancer-causing radioactivity.

- Limit the inventory of radionuclides with the greatest potential to harm streams, fish, and people because they can travel, accumulate, or cause cancer or other health effects. Attachment 1 presents examples of radionuclides that may need inventory limits.

- The ROD must require a comprehensive program that monitors radionuclides and other hazardous contaminants in fish and uses CERCLA guidance to evaluate potential risks to people consuming fish caught downstream. The program must include ways to inform the public of any risks, including posting streams with unacceptable risks and contingencies for corrective action including, but not limited to, closing EMDF if necessary to decrease unacceptable risks.
- The ROD must state that OREM will track the amounts of all radionuclides identified in waste profiles and placed in the proposed EMDF landfill, including radioactive decay progeny, to determine the total amount of radioactivity in the EMDF at any time and to document the total inventory upon closure.

- Specify that OREM will not place Greater-Than-Class-C waste in the proposed EMDF.

- Involve TDEC & EPA in defining and approving operations-based WAC.

- Include language in the ROD obligating OREM to:

    - Submit a primary document under the FFA that lays out requirements for an independent WAC compliance plan for approval by EPA & TDEC; and

    - Implement independent WAC compliance with regulatory oversight, including establishment of an independent WAC compliance team with authority to direct waste characterization efforts and waste profile development.

Attachment 1

Preliminary List of Radionuclides Needing Inventory Limits at
the Proposed Environmental Management Disposal Facility (EMDF)
Tennessee Department of Environment & Conservation (TDEC)

The Tennessee Department of Environment & Conservation (TDEC) developed a list of radionuclides (Table 1) that need inventory limits as the U.S. Department of Energy (DOE) sets waste acceptance criteria (WAC) for the proposed Environmental Management Disposal Facility (EMDF) in Oak Ridge. TDEC prepared the list because of the need to protect people who eat fish caught near the proposed EMDF site *in the future*, as required by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).[10,11] The proposed EMDF site, also known as the Central Bear Creek Valley site, lies on Bear Creek downstream of two existing disposal sites—Bear Creek Burial Grounds (BCBG) and the Environmental Management Waste Management Facility (EMWMF).

TDEC applied the following assumptions in developing the list in Table 1.

- EMDF waste cells will occupy 23 acres, as projected in the EMDF D5 remedial investigation/feasibility study (RI/FS).
- The EMDF landfill returns to a more natural condition in the future—e.g., 250, 1,000, or 10,000 years.
- The natural recharge rate is 9.4 inches per year, the average measured in the Poplar Creek watershed near Oak Ridge and published by the U.S. Geological Survey (USGS).[12]
- Given that Bear Creek is designated for recreational use, stream flow was estimated using USGS StreamStats[13] based on the "*30-day minimum five year recurrence interval,*" per Tennessee Comp. R. & Regs. 0400-40-03-.05(4).
- A person eats 24 ounces (oz), which is three 8-oz or four 6-oz servings, of fish caught downstream each month for 26 years.
- An acceptable excess lifetime cancer (ELCR) of "*10^{-5} risk level is used for all carcinogenic pollutants,*" per Tennessee Comp. R. & Regs. Rule 0400-40-03-.03(4)(j).
- Bioconcentration factors (BCF) are the geometric means of International Atomic Energy Agency (IAEA) / International Union of Radioecologists (IUR) values, which are updates to the IAEA BCF values in the Oak Ridge National Lab (ORNL) Risk Assessment Information

[10] While it is reasonable to assume members of the public are unlikely to access controlled portions of the Oak Ridge Reservation upstream of Highway 95 on a routine basis during landfill operations, that assumption is neither reasonable nor consistent with CERCLA risk assessment guidance for evaluating periods after DOE no longer controls the land use.

[11] CERCLA is a set of laws Congress created to clean up the most polluted sites in the country. Commonly known as "Superfund," CERCLA is the Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended by the Superfund Amendments and Reauthorization Act (SARA) of 1986.

[12] Hoos, A.B., 1990, Recharge Rates and Aquifer Hydraulic Characteristics for Selected Drainage Basins in Middle and East Tennessee: U.S. Geological Survey Water-Resources Investigations Report 90-4015, 34 p. (available at https://pubs.usgs.gov/wri/wri90-4015/pdf/wrir_90-4015.pdf)

[13] Available at https://streamstats.usgs.gov/ss/.

System (RAIS) and U.S. Environmental Protection Agency (EPA) Preliminary Remediation Goals for Radionuclides (PRG) calculator.[14]

TDEC's evaluation proceeded as follows.

- Evaluate radionuclides the EMDF D5 RI/FS projects will have non-negligible inventories.
- Estimate releases (by isotope) that may cause a 10^{-5} excess cancer risk if the landfill returns to a more natural condition in 250, 1,000, or 10,000 years.
- Assess fish consumption risk using the EPA radionuclide PRG calculator with default assumptions, except the fish ingestion rate, risk level, and BCF values described above.
- Calculate average initial radionuclide concentrations in the waste that would decay to concentrations associated with the 10^{-5} risk level in 250, 1,000, and 10,000 years.
- Identify long-half-life radionuclides with high partition coefficients (K_d) and BCF values to identify isotopes with "uncertain" long-term fish consumption risks.
- Review the radionuclide list in the D3 RI/FS to determine if additional long-half-life radionuclides with elevated BCF and K_d values need further evaluation.
- Determine radionuclides of potential concern during the 1,000- to 10,000-year period by comparing calculated inventories (in curies) that may be disposed of without exceeding the 10^{-5} risk level with the projected inventory from the D5 RI/FS.[15]
- Repeat evaluation as if EMWMF and BCBG would be the only disposal sites in Bear Creek Valley.

Table 1 presents the radionuclides of potential concern, including long-half-life isotopes that may be present at EMWMF and BCBG in sufficient quantities to warrant more thorough evaluation. These isotopes need additional evaluation to determine whether the combined inventories of EMDF, BCBG, and a future EMDF would exceed the 10^{-5} risk level, which would violate Tennessee Comp. R. & Regs. 0400-40-03-.03(4)(j) and, thus, the CERCLA requirement for overall protection of human health.

TDEC also evaluated K_d and BCF values to identify isotopes with "uncertain" long-term fish consumption risks. Radionuclides with high K_d values tend to sorb to sediment and organic matter. They move downstream with sediment load and may stay in the creek longer. BCF values indicate bioconcentration in the food web. BCF values for some radionuclides vary by orders of magnitude, so TDEC used central-tendency values (geometric means) where available.

TDEC's evaluation retained several radionuclides that the D5 RI/FS screened out. For example, the D5 RI/FS screened out chlorine-36 (Cl-36) and potassium-40 (K-40) due to negligible inventory, yet both isotopes have been detected in EMWMF discharge at the V-Weir. Further, the K-40

[14] Available at https://epa-prgs.ornl.gov/cgi-bin/radionuclides/rprg_search.
[15] For many of the radionuclides, half-lives are long enough to prevent significant change in the inventory associated with the 10-5 risk level from 250 to 10,000 years.

detection limit for V-Weir analyses is insufficient to evaluate risk. Neither Cl-36 nor K-40 requires a large inventory to pose potential risk to people eating fish caught downstream.

The D5 RI/FS also screened out Cesium-135 (Cs-135) as having a negligible inventory. The inventory may appear negligible due to the challenge of analyzing Cs-135 in the laboratory. Table 1 includes Cs-135 because it is a fission product formed with Cs-137, which is in the projected inventory. If there is a substantial quantity of Cs-137 in the waste, there may also be a substantial quantity of Cs-135. Moreover, Cs-135 has a half-life of 2,300,000 years, much longer than the 30.17 half-life of Cs-137. Cs-134, Cs-135, and Cs-137 have relatively high K_d and BCF values and are important considerations for the downstream fishing exposure pathway.

The list of radionuclides in Table 1 is not a complete list of isotopes that require monitoring during the operational and post-closure periods. Table 1 excludes most radionuclides with shorter half-lives because landfill wastewater treatment, operations, and post-closure care should protect people fishing downstream during the short term. The list of radionuclides in Table 1 focuses primarily on (1) long-half-life radionuclides (1,000 to 10,000-year evaluation), (2) radionuclides with potentially large inventories, and (3) radionuclides identified as concerns due to combinations of BCF, K_d, and toxicity. Cs-135 was also added due to characterization uncertainty and its potential impact on fish consumption risk.

Table 1 lists the radionuclides that appear to need inventory limits at the proposed EMDF landfill to protect people who eat fish caught downstream after DOE no longer controls the area. There should be a means to add WAC requirements, including inventory limits, for additional radionuclides, if needed. For example, radionuclides projected to have negligible inventories may be disposed of in greater quantities than originally anticipated, and additional radionuclides may be found to be significant at ORNL or other waste sources.

Attachment 1

Preliminary List of Radionuclides Needing Inventory Limits at
the Proposed Environmental Management Disposal Facility (EMDF)
Tennessee Department of Environment & Conservation (TDEC)

Isotope	**Half-life (years)**	**Seven Half-lives (years)**	**1,000- to 10,000-year Evaluation**	**Comment**
Americium-241 (Am-241)	432.2	3,025	Potential Concern	
Americium-243 (Am-243)	7,370	51,590	Potential Concern	
Carbon-14 (C-14)	5,700	39,900	Potential Concern	(1) Detection limits for C-14 in CY2019 at EMWMF V-Weir are often insufficient to determine risk at 10^{-5}. (2) Evaluate C-14 at EMWMF and combined impact with proposed EMDF on risk from consuming fish caught in Bear Creek. EMDF inventory should incorporate C-14 disposed of at EMWMF.
Chlorine-36 (Cl-36)	301,000	2,107,000	Potential Concern	Periodically measured in discharge at EMWMF V-Weir
Curium-245 (Cm-245)	8,500	59,500	Uncertain	High K_d (9,000 L/kg); low BCF (0.24 L/kg) BCF based on limited data (7 samples in 1 reference)
Curium-246 (Cm-246)	4,760	33,320	Uncertain	High K_d (9,000 L/kg); low BCF (0.24 L/kg) BCF based on limited data (7 samples in 1 reference)
Curium-247 (Cm-247)	15,600,000	109,200,000	Uncertain	High K_d (9,000 L/kg); low BCF (0.24 L/kg) BCF based on limited data (7 samples in 1 reference)
Cobalt-60 (Co-60)	5.3	37	No	Large inventory released at ORNL and high activity level mean inventory limits may be necessary to ensure landfill wastewater discharges can be treated to protective discharge limits
Cesium-135 (Cs-135)	2,300,000	16,100,000	Uncertain	Hard to measure fission product generated with Cs-134 and Cs-137; ORNL inventory needs re-evaluation with analytical methods sufficient to measure Cs-135 and Cs-137 atomic ratios; High BCF (1700 L/kg) and K_d (1200 L/kg)
Cesium-137 (Cs-137)	30.2	211	No	Large inventory released at ORNL High BCF (1700 L/kg) and K_d (1200 L/kg)

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Tennessee Department of Environment & Conservation (TDEC)

Isotope	**Half-life (years)**	**Seven Half-lives (years)**	**1,000- to 10,000-year Evaluation**	**Comment**
Gadolinium-148 (Gd-148)	74.6	522.2	No	D3 RI/FS Radionuclide list; High BCF and K_d: Fish BCF of 1200 L/kg with K_d of 650 L/kg
Gadolinium-150 (Gd-150)	1,790,000	12,530,000	Uncertain	D3 RI/FS Radionuclide list; High BCF and K_d: Fish BCF of 1200 L/kg with K_d of 650 L/kg
Gadolinium-152 (Gd-152)	1.08e+14	7.56E+14	Uncertain	EMDF D3 RI/FS Radionuclide list; High BCF and K_d: Fish BCF of 1200 L/kg with K_d of 650 L/kg
Iodine-129 (I-129)	15,700,000	109,900,000	Potential Concern	
Potassium-40 (K-40)	1,251,000,000	8,757,000,000	Potential Concern	Detection limits for K-40 in CY2019 at EMWMF V-Weir are often insufficient to determine risk at 10^{-5}
Lead-210 (Pb-210)	22.2	155	As Progeny	Lead-210 will decay in a few hundred years. However, it is toxic and is progeny of Ra-226, Th-230, U-234, Th-234, and U-238.
Nickel-63 (Ni-63)	100.1	701	No	Potential for a large inventory, and 700 years is a long time.
Neptunium-237 (Np-237)	2,144,000	15,008,000	Potential Concern	
Plutonium-238 (Pu-238)	87.7	614	No	Potential for a large inventory, and 600 years is a long time.
Plutonium-239 (Pu-239)	24,110	168,770	Potential Concern	
Plutonium-240 (Pu-240)	6,564	45,948	Potential Concern	
Plutonium-242 (Pu-242)	375,000	2,625,000	Potential Concern	
Plutonium-244 (Pu-244)	80,000,000	560,000,000	Potential Concern	
Radium-226 (Ra-226)	1,600	11,200	Potential Concern	
Selenium-79 (Se-79)	295,000	2,065,000	Uncertain	D5 RI/FS screened out due to negligible inventory; High BCF and K_d: BCF (1,600 L/kg) and K_d (200 – 1,000 L/kg)
Tin-126 (Sn-126)	230,000	1,610,000	Uncertain	D5 RI/FS screened out due to negligible inventory; High BCF and K_d: BCF (400 L/kg) K_d (1,600 L/kg)

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Isotope	Half-life (years)	Seven Half-lives (years)	1,000- to 10,000-year Evaluation	Comment
Strontium-90 (Sr-90)	28.79	202	No	Large inventory released at ORNL
Technetium-99 (Tc-99)	**211,100**	1,477,700	Potential Concern	Need to evaluate Tc-99 at EMWMF and combined impact with proposed EMDF on risk from consuming fish caught in Bear Creek. EMDF inventory should incorporate Tc-99 disposed of at EMWMF.
Thorium-229 (Th-229)	7,340	51,380	Uncertain	K_d of 3,300 L/kg and BCF of 120 L/kg
Thorium-230 (Th-230)	75,380	527,660	Uncertain	K_d of 3,300 L/kg and BCF of 120 L/kg
Thorium-232 (Th-232)	1.405E+10	9.84E+10	Potential Concern	K_d of 3,300 L/kg and BCF of 120 L/kg
Thorium-234 (Th-234)	0.066	0.46	Potential Concern	Secular equilibrium with U-238
Uranium-233 (U-233)	159,200	1,114,400	Potential Concern	
Uranium-234 (U-234)	245,500	1,718,500	Potential Concern	Need to evaluate U-234 at EMWMF and combined impact of EMWMF and proposed EMDF on risk from consuming fish caught in Bear Creek. EMDF inventory should incorporate U-234 disposed of at EMWMF.
Uranium-235 (U-235)	704,000,000	4,928,000,000	Potential Concern	
Uranium-236 (U-236)	23,420,000	163,940,000	Potential Concern	

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Isotope	**Half-life (years)**	**Seven Half-lives (years)**	**1,000- to 10,000-year Evaluation**	**Comment**
Uranium-238 (U-238)	4.468E+09	3.1E+10	Potential Concern	1) Determine U-238 inventory disposed of at EMWMF and BCBG. 2) Subtract that inventory from the U-238 inventory that may be disposed of at proposed EMDF without posing unacceptable risk through the fish consumption pathway. 3) Alternatively, the EMDF ROD could include a requirement and timeline for cleanup of BCBG.

BCBG - Bear Creek Burial Grounds
BCF - bioconcentration factor
EMDF - Environmental Management Disposal Facility
EMWMF - Environmental Management Waste Management Facility
Kd - partition (or distribution) coefficient
L/kg - liters per kilogram
ORNL - Oak Ridge National Laboratory
RI/FS - Remedial Investigation/Feasibility Study
ROD - Record of Decision

From:	Brad Stephenson
To:	Dennis Mayton (dennis.mayton@orem.doe.gov)
Cc:	DePaoli, Susan (CONTR); Carl Froede (froede.carl@epa.gov); Richards, Jon M.; Michael D. Higgins
Subject:	EMDF Draft WAC: TDEC Preliminary Review
Date:	Monday, June 15, 2020 3:01:00 PM
Attachments:	73212_EMDF_WAC_TDEC_Review_06_15_2020.pdf image001.png

Dennis and Susan,

Good afternoon. The attached file presents TDEC's preliminary review of draft Waste Acceptance Criteria (WAC) information. DOE shared draft WAC information in an excerpt from the forthcoming EMDF ROD and requested feedback from TDEC.

We look forward to discussing this review and working with DOE and EPA to develop protective WAC in the ROD and ensure compliant waste disposal.

TN Department of Environment & Conservation

J. Brad Stephenson, PG
Division of Remediation | Oak Ridge
761 Emory Valley Rd
Oak Ridge, TN 37830
p. 865-220-6587
f. 865-482-1835
Brad.Stephenson@tn.gov
tn.gov/environment

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