Independent Scenario Analysis of the Hecla Durita Heap Leaching Site

Final Report

Attachment BB

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Abstract

In March 2016, the Colorado Department of Public Health and Environment (CDPHE, 2016) submitted an updated Completion Review Report (CRR) to the U.S. Nuclear Regulatory Commission (NRC) for the Title II Durita uranium heap leaching site in Montrose County, Colorado. NRC staff comments on earlier drafts of the CRR and other site documents indicated concerns with the (i) early termination in 1998 of the groundwater detection monitoring program, (ii) adequacy of the monitoring network for early detection of contaminant releases, and (iii) reliance on the thin Mancos shale layer to act as a barrier even though the potential presence of a transmissive subsurface fracture network (NRC 2005a, b; NRC, 2014) extending to the designated uppermost aquifer could not be discounted.

This groundwater monitoring issue is addressed by NRC staff in this report in two regards: (i) evaluation of the designated uppermost aquifer and the hydraulic connection with the aquifer at greater depths that is currently being used for water supply in the area, and (ii) evaluation of the consequences of a potential release of contaminants to the uppermost aquifer using conservative assumptions in a scenario analysis. To address the first issue, the aquifer assessment considers site features indicative of groundwater quality and production capability of the designated uppermost aquifer in light of the regulatory definition of an aquifer in Title 10 of the *Code of Federal Regulations* (10 CFR) 40, and considers site information that indicates no hydraulic connection between that aquifer and deeper aquifers. For the second element, the NRC scenario analysis, a site conceptual model is developed for two potential transport pathways to the designated uppermost aquifer. The site conceptual model incorporates geochemical characteristics for the site. The first transport pathway follows vertical fractures in the bedrock to the uppermost designated aquifer and then lateral migration to the site boundary. The second transport pathway follows the unconsolidated sediment and bedrock interface to the site boundary and vertical migration in the bedrock to the designated uppermost aquifer. The modeling is performed using RESRAD-OFFSITE version 3.2 [NUREG/CR-7189 (NRC, 2015)] with results reported as activity concentrations in the designated uppermost aquifer reaching the site boundary. Uncertainty is addressed by implementation of the two scenario pathways and conservative assumptions and inputs for each scenario. In addition, sensitivity analyses are performed for selected important input parameters.

The NRC staff considers the results presented herein informative for its evaluation, especially in light of the absence of some site-specific subsurface data. Even with the combination of several worst case assumptions, conservative inputs, and sensitivity analyses for several important parameters, the modeling results suggest no issues that would constrain the NRC staff from concluding the site's features will be effective in controlling radiological hazards for 1,000 years, the performance period specified 10 CFR Part 40, Appendix A, Criterion 6(1). The staff recognizes that the modeling results do not reflect a most likely or expected level of contamination reaching the site boundary because of the extensive use of conservative inputs and assumptions. The staff concludes that based on the site features, aquifer assessment, and scenario analyses, that with respect to groundwater, the Durita site would provide reasonable assurance of control of radiological hazards effective for 1,000 years.

1   Introduction

An updated Completion Review Report (CRR) developed by the Colorado Department of Public Health and Environment (CDPHE) (CDPHE, 2016) was provided to the U.S. Nuclear Regulatory Commission (NRC) for the Title II Durita uranium heap leaching site in Montrose County, Colorado. Earlier NRC reviews (e.g., NRC, 2005a,b; NRC, 2007; NRC, 2014) of draft CRRs and supporting site documents indicated that the absence of detected constituents in the groundwater monitoring wells was not thought to be sufficient for an affirmative conclusion of safety because of the early termination of the detection monitoring program and the possible inadequacy of the monitoring network considering site hydrogeological conditions. There has been no groundwater monitoring at the site since 1998 and a request for license termination is currently pending. A detection groundwater monitoring program is required by Title 10 of the Code of Federal Regulations (10 CFR) 40, Appendix A, Criterion 7(A) during the compliance period, which is defined as the period when the groundwater protection standards are set and ends when the license is terminated. The inadequacy of the monitoring network reflects on plausible subsurface site characteristics that may include the presence of transmissive fracture networks in the shale bedrock unit overlying the thin sandstone layers of the designated uppermost aquifer. This report does not address erosion or settling of the engineered covers, nor gaseous emission of radon-222 from the impoundments. Groundwater is the sole focus of this report.

Site History

The license for the Durita site was issued in 1976 to Ranchers Exploration and Development Corporation. In 1977, the facility was constructed and began operations as a secondary-extraction heap leach facility that recovered uranium and vanadium from mill tailings that had originally been processed through a mill in Naturita, Colorado. The leach tanks and evaporation ponds at Durita were located in an area with subcropping Mancos Shale bedrock and were constructed with compacted clay liners. Tailings were transported to the Durita site, placed into one of three heap leach tanks, and dilute sulfuric acid solution was applied. The solution leached uranium and vanadium that was then transferred to an onsite extraction plant where uranium and vanadium were recovered by ion exchange and solvent extraction. Liquid waste was stored in six onsite evaporation ponds to the north, and downgradient, of the heap leach tanks. Figure 1 illustrates the layout of operations at the Durita site. Operations at the site ceased in May 1979.

In 1984, Hecla Mining Company merged with Ranchers Exploration and Development and became the licensee of the Durita site. From the end of operations in 1979 to 1993, site activities consisted of licensee custodial care, monitoring, and some decontamination. Hecla decommissioned the site during the period 1993 through 1998. The evaporation ponds were excavated and geochemically stabilized in an onsite engineered closure cell; some demolition debris and radioactively contaminated soil were also disposed in the cell. The closure cell included compacted clay layers as a liner and in the cover system. The tailings in the leach tanks were drained and the collection pipes at the base of the tanks were plugged with
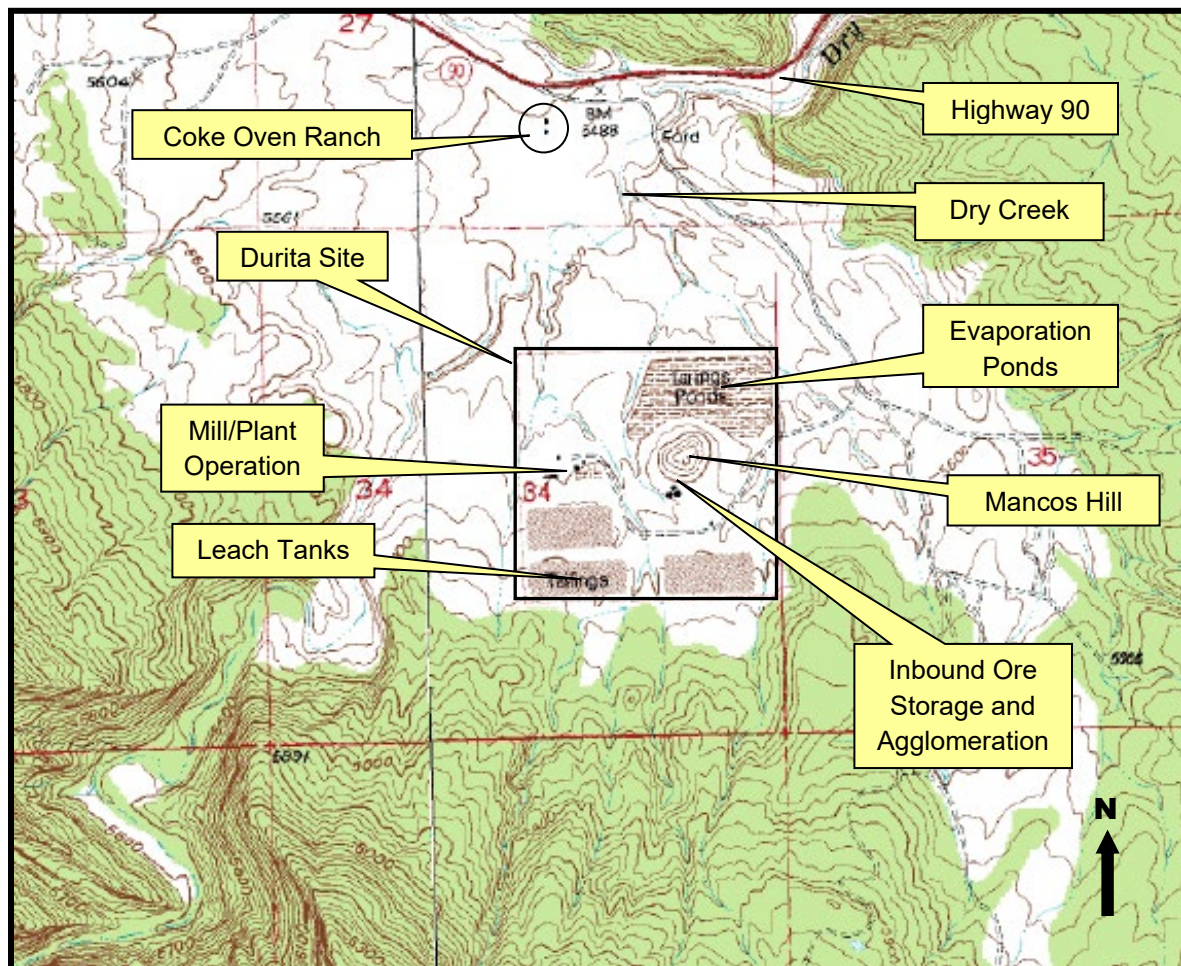
Figure 1. Site Features and physiography of surrounding area. Closest producing well to the Durita Site is at the Coke Oven Ranch. Base topographic map at 1:50,000 scale downloaded from Topozone™ on 07/17/2006.

concrete. The leach tanks were covered with an engineered earthen radon barrier that included a compacted clay layer, the top surface re-vegetated, and a rock layer added to prevent erosion on the out slopes. During reclamation, some mill debris was disposed by onsite burial in the out slopes and toes of the leach tanks. Surface reclamation activities at the site, in accordance with the plan that was approved by CDPHE in 1993, were completed in 1999 and approved by Colorado in the CRR (CDPHE, 2016).

In 1991, a second set of monitoring wells replaced the first set to rectify well construction and screen-depth issues of the earlier set. The second set of monitoring wells was screened exclusively on the thin sand layers near the base of Mancos Shale and at the transition to the Dakota Formation, which CDPHE collectively designated as the uppermost aquifer. The only producing well in the area is developed in the aquifer of the lower Dakota Formation, which is not at risk for contamination from the Durita site. CDPHE approved termination of the

groundwater monitoring program in 1998 and the monitoring wells were plugged in 2002. CDPHE submitted a draft CRR in 2004 for NRC review.

Discussions between CDPHE and the NRC staff from 2004 to the present revolved around the issue of groundwater monitoring and potential contamination of the designated uppermost aquifer. In an earlier round of modeling (NRC, 2007; CDPHE, 2005a), CDPHE and the NRC staff have both cited features at the site that may reduce the possibility and consequences of a release of radionuclides from the leach tanks or closure cell to reach the designated uppermost saturated layer. However, the results differed between NRC and CDPHE mostly due to the differing assessment of the site-specific support for the effectiveness of the thin overlying shale layer to act as a low permeability barrier to contaminant releases from the leach tanks. Without appropriately placed monitoring wells near the source, the plausible presence of through going fractures in the shale that would act as conduits for contaminant releases to the designated uppermost aquifer could not be discounted. Furthermore, the detection wells were located at distances from the source area that were well beyond migration distances estimated from (non-fractured rock) flow and transport conditions at the site described in the CRR (CDPHE, 2016).

This Report

Groundwater monitoring is addressed in this report in two approaches: (i) assess the designated uppermost aquifer at the site and the potential hydraulic connection with a lower aquifer that is being used for water supply in the area, and (ii) evaluate the consequence of a potential release of contaminants to the CDPHE designated uppermost aquifer using conservative assumptions in a scenario analysis. The consequence modeling addresses the uncertainty of the effectiveness of the overlying shale layer to act as a barrier to a release from the leach tanks.

For the first approach, the designation of this near-surface saturated horizon does not fit the definition of an aquifer in 10 CFR Part 40. The designated uppermost aquifer for compliance is in the lower portion of the Mancos Shale, and is referred to as the designated uppermost saturated layer in this report because it does not fit the 10 CFR 40 Appendix A definition of an aquifer. A detailed discussion is provided in Section 3, Designated Uppermost Aquifer. Water use in the area taps an aquifer in the lower Dakota Formation (below the Mancos) that is not hydraulically connected to the thin near-surface saturated layer designated as the uppermost aquifer by CDPHE. The nearest well, located a half mile to the north at the Coke Oven Ranch (Figure 1), taps an aquifer in the lower portion of the Dakota Formation and is not at risk for contamination from the Durita site based on groundwater flow directions that are constrained by structural geology of the area. In addition, distinct geochemical conditions between the shallow saturated horizons at the site and in the deeper horizons of the lower Dakota Formation indicate the two are not in hydraulic connection. Context and detailed discussion of aquifers is provided in Section 2, Site Characteristics, and assessment of aquifers considering the regulatory requirements is provided in Section 3, Designated Uppermost Aquifer.

For the second approach, Sections 4 through 7 of this report describe conceptual model development, modeling inputs, and results. The hydrogeology of the Durita site is assessed

based on available documents to evaluate the consequence of a hypothetical release of radionuclides from the leach tanks and closure cell. The hypothetical release encompasses the possibility of past releases and potential future releases. A conceptual model is developed for two potential pathways for radionuclide contaminants to migrate to a hypothetical well placed at the compliance boundary (referred to as the fence line or site boundary) in the designated uppermost saturated layer. Concentrations in the designated uppermost saturated layer at the compliance boundary are calculated using the flow and transport components of RESRAD-OFFSITE version 3.2 [NUREG/CR-7189 (NRC, 2015)].

Previously, CDPHE (2005a) and NRC (2007) modeled the consequence of potential releases using travel time calculations. Travel time calculations use an estimate of particle velocity based on Darcy's Law and effective porosity, instantaneous release of the contaminant, retardation, and distance to the compliance location. Retardation is readily incorporated into the travel time analysis using a sorption coefficient, which may also be called a distribution coefficient. The CDPHE (2005a) travel time estimate for contaminant transport to the site boundary was 22 million years, with a safety factor adjustment to 22,000 years. Using ranges for sorption, permeability, and distance to compliance location, NRC (2007) found some combinations of the uncertain parameters led to travel times of less than 1,000 years.

Modeling the consequences of potential releases in this report goes beyond the travel time calculations presented in CDPHE (2005a) and NRC (2007). The calculations in this report include a linear release rate from a source volume, radionuclide decay and ingrowth, and transport through the multiple geologic layers. The calculations consider geochemical conditions and radionuclide sorption for each layer. Extensive discussions of the source term and geochemical behavior of potential releases to the groundwater system are included in this report because they have not been previously described in detail. Some information from the Naturita operations and site are incorporated into this analysis as relevant; tailings material from the Naturita milling operation were loaded into the leach tanks at Durita for additional uranium extraction. Estimation of radionuclide concentrations in the leach tank required analysis of geochemical conditions in the ore and processing efficiencies at Naturita and Durita.

Where site-specific information is not available for inputs, conservative choices and sensitivity analyses are utilized. The staff also uses available information and its technical judgment to develop the scenarios and input values that are conservative to performance when incorporated into models where site-specific is not available. The NRC staff considers this approach valid for a study used to help judge safety, even though it may not necessarily reflect a most likely condition or feature of the site. The modeling includes sensitivity analyses for the (i) uranium distribution coefficient, (ii) conceptual model for transport pathways, and (iii) performance of the engineered barriers. For the latter, the engineered tailings covers and liners are conservatively assumed to provide limited performance; i.e., the compacted clay of the engineered cover is assumed to perform no better than would the natural sediments in the area and the engineered liner is assumed to fail to various degrees. In summary, the NRC staff modeling uses a combination of several worst case assumptions, conservative inputs, and sensitivity analyses

for several important parameters to provide information used to judge the site's effectiveness in controlling radiological hazards for 1,000 years.

2   Site Features and Conceptual Model

The Durita site has features that are advantageous for containment of a potential release and reduction of transport away from the site, but also has features and indications of the potential for contaminants to migrate to downstream environments. Features and conditions at the site that affect potential flux through the leach tanks and transport to the accessible environment are described in this section. They include the site's geological context, flux of water through the groundwater system, monitoring of the groundwater, and cover and liner performance.

The descriptions in this section are not intended to replace those in CDPHE or site operator documents, but rather are intended to provide enough context to understand discussions of the modeling scenarios and inputs in subsequent sections.

2.1   Climate and Infiltration

The Durita site has a semi-arid climate for which the CRR (CDPHE, 2016) reports annual precipitation of 12 inches (30 cm). This is consistent with WRCC (2017) data indicating approximately 12.5 inches (32 cm) annual precipitation based on 1960-2016 data for Uravan, which is 10 miles (16 km) north of the Durita site. Natural vegetation in in the vicinity of the site is dominantly sage bush and grasses, with pinyon and juniper on slopes and ridges.

In semi-arid areas of the western U.S., runoff and evapotranspiration significantly reduce the amount of precipitation that results in groundwater recharge, which is infiltrating water that percolates below the rooting zone. Potential evaporation for the Durita site was estimated to be 49 inches/yr (1.2 m/yr) (Four Corners ER, 1977); this was reported as evaporation in CDPHE (2016) and as net evaporation in CDPHE (2007). The NRC staff notes that potential evaporation estimates cannot be compared directly to precipitation to determine if or how much recharge occurs. CDPHE (2007, p.5) stated that "most, if not all," infiltrated water is eliminated by evaporation and transpiration, and provided as a basis that evaporation is three times the precipitation rate. Pan evaporation measurements reported in the Four Corners ER (1977) reflect operation of the evaporation ponds, and cannot be used directly to infer negligible recharge through the unconsolidated sediments or Mancos Shale. For the leach tanks and closure cell directly, CDPHE (2016, p.14) reported "percolation/infiltration" estimates of 0.00043 inch/yr (0.0011 cm/yr) in the closure cell and 0.00103 inch/yr (0.0026 cm/yr) in the leach tanks based on 1993 calculations using the HELP v3.3 code (Hydrologic Evaluation of Landfill Performance, Schroeder, et al., 1994). Using 1997 calculations, CDPHE (2016, p.19) reported "percolation/infiltration" estimates of 0.0011 inch/yr (0.0028 cm/yr) in the closure cell and 0.0019 inch/yr (0.0048 cm/yr) in the leach tanks using the same code as the 1993 calculations. No supporting information was provided for either set of estimates. These infiltration rates for the

leach tanks and closure cell are a small fraction of one percent of annual precipitation (0.016 to 0.0036 percent).

The typical hydrological regime in semi-arid areas of the western U.S. with bushes and sparse grasses growing in unconsolidated sediments covering a shallow bedrock is for infiltration to be highly variable year-to-year with a prominent seasonal difference (Meixner et al., 2016; Hogan et al., 2013; Stonestrom et al., 2007; Scanlon et al., 2005). In these areas, significant recharge passing below the root zone may not occur every year. Wet winters, especially during El Niño years, with long duration but low precipitation intensities from frontal weather systems generally lead to more recharge than that associated with high intensity convective summer weather systems where the runoff percentage is large. Bedinger (1987) summarized results of analyses across the arid western U.S. that span a range of 0.01 to 25 percent of precipitation that becomes recharge depending on local conditions and hydrogeology. Areas with clayey soils and extensive vegetation fall at the lower end, and areas with sandy/gravelly soils and sparse vegetation fall at the upper end of the recharge range. Annual average recharge over the long-term for environments similar to Durita in the western U.S. is approximately 1 to 2 percent of precipitation. This range of 1 to 2 percent is significantly larger that the range estimated by the licensee, noted in the paragraph above, of 0.016 to 0.0036 percent. Staff notes that values much higher than 2 percent lead to inconsistencies in the groundwater flow balance at Durita, given reasonable ranges of permeability, measured gradients, thickness of the saturated hydrogeological unit, and hydrological conditions observed at the site.

The NRC staff conservatively assumes that infiltration and recharge into the leach tank tailings over the long-term is the same as through the natural unconsolidated sediments. This assumption means that the compacted clay layer would not perform as designed, but rather would act no differently than the more permeable natural sediments in the area. In the Site Engineered Features subsection of Section 2, the NRC staff notes that field observations of long-term performance of covers indicate that they may revert to the hydrological characteristics of the natural sediments at the sites (e.g., Benson et al. 2011). Uncertainty in infiltration and percolation is treated with sensitivity analysis in Section 6.

2.2 Hydrogeology

The geological units at the site include unconsolidated sediments overlying bedrock units that include remnants of the lower Mancos Shale conformably overlying the Dakota Formation. The Mancos Shale includes shales, sandstone, and carbonate layers. CDPHE (2016, Table 7.1) refers to Units A, B, and C for the hydrogeological layers: Unit A is the weathered bedrock and unconsolidated sediments, Unit B is the unweathered Mancos Shale, and Unit C is the lowermost Mancos extending to the upper Dakota Formation, which also contains shale, sand, and carbonate layers.

Unconsolidated Sediments

Eolian sediments are mapped as the surficial material across much of the Durita site (Fox 1982, Figure 17). The unconsolidated sediments below the surface are comprised of a mixture of colluvium, alluvium, and eolian deposits. The thickness of the sediments varies across the site from 0 to >20 ft (0 to >6.1 m) based on borehole logs (CDPHE, 2015); in addition, the thicknesses have been modified by site activities. The unconsolidated sediments are more permeable than the underlying unweathered Mancos Shale bedrock.

The weathered portion of the bedrock is hydraulically connected to groundwater in the unconsolidated sediments, and as such, is lumped with the unconsolidated sediments in hydrogeological Unit A. The upper portion of the bedrock is generally weathered where exposed and in borehole logs, though the weathered thickness is not well constrained. Based on borehole descriptions (CDPHE, 2015), the thickness of the weathered Mancos Shale ranges from 3 to 12.5 ft (1 to 3.8 m) with an average of 6.2 ft (1.9 m). The weathered bedrock is more permeable than the unweathered bedrock (CDPHE, 2016) due to fracturing and chemical alteration related to near-surface processes.

Saturated horizons were not found in boreholes in the unconsolidated sediments on the south side of Dry Creek – the Durita site is on the south side of Dry Creek. Fox (1982), however, found zones of saturation in the unconsolidated sediments adjacent to and on the north side of Dry Creek. They postulated a seasonal fluctuation in the water table with recharge in the spring and summer and declining water levels through the fall and winter. Fox (1982), however, states that there is no current use of the shallow groundwater (industrial, agricultural, residential, or otherwise) in the alluvial aquifer on the north side of Dry Creek, nor is there likely sufficient water quality or quantity to support use for the groundwater. The presence of saturated zones in the unconsolidated sediments on the north side of Dry Creek suggests the possibility of locally saturated areas may occur on the south side, albeit in lower amounts that may only periodically flow. Based on the topographic slopes, saturated horizons are less likely to occur on the hillslope of the south side of the channel (nearer the Durita site) than on the flat valley bottom of the north side of the channel. Additional possible support for the occurrence of at least intermittent flow along the interface is the mineral alteration at the bedrock/sediment interface exposure on the south side of the Dry Creek channel; although the mineral alteration may be a relic of the last pluvial climate period more than 10,000 years ago.

Where unconsolidated sediments overlie the Mancos bedrock at the Durita site, infiltration into the unconsolidated sediments likely occurs, but may be highly variable temporally. Flow along the base of the weathered bedrock and unconsolidated sediment, if it occurs, would be both spatially and seasonally-to-annually variable. Borehole geologic data indicate the presence of a paleo-drainage surface that would channelize flow along the bedrock/sediment interface. Analysis of the elevation and thickness data from borehole and exploratory test hole logs found in Hecla (1991) and Fox (1977) provide an indication of two paleochannels along the bedrock/sediment interface. These paleochannels approximately correspond to the two present-day drainage channels crossing the site (see Figure 2). The more prominent

paleochannel crosses the site from the south-central boundary in a north-northwest direction toward monitoring well MW-4. This channel would capture flow from the area of the two larger volume leach tanks (LT-201 and LT-202) and part of the flow from the smaller, third leach tank (LT203). The less well defined paleochannel passing to the east of Mancos Hill would capture flow northward only from a portion of leach tank LT-203. Consistent with this conceptualization of flow is the Four Corners ER (1977) suggestion that shallow groundwater flows through the unconsolidated sediments below arroyos (i.e., subsurface baseflow along washes or channels),
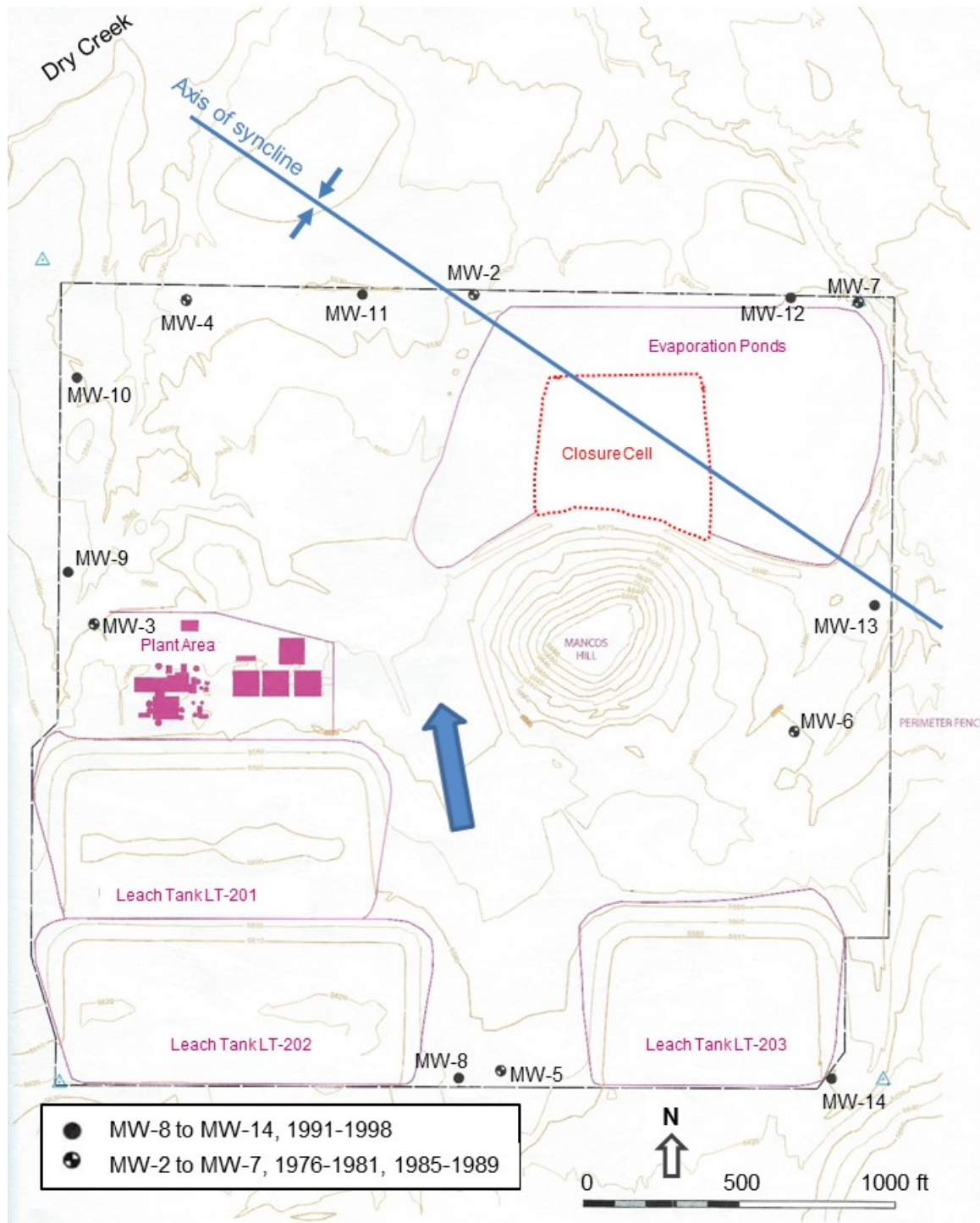
Figure 2. Layout of Durita Site with monitoring well (MW) locations and topography, modified from Hecla (2007). Groundwater flow direction (blue arrow) in the Mancos Shale is generally northward; the large blue arrow also marks the main channel through the site. Approximate location of major axis of Coke Oven Syncline is derived from Figure 9 of Fox (1982). Gap in years monitored for MW-2 to MW-7 reflects a loss of records (CDPHE, 2016, p.89). [3.3 ft = 1 m].

especially after periods of runoff. CDPHE (2016, Figure 7.3) referred to this paleo-topographic surface as the aquitard surface in the context of the Unit C confined groundwater layer, and mentions the possibility of unconfined flow above the aquitard.

This possible intermittent presence of shallow unconfined groundwater flow leads to the transport pathway in the modeling described in Sections 4 and 6. Support for an unconfined groundwater transport pathway in the unconsolidated sediments and weathered bedrock is provided by a deduced spill from the area of the former Plant Building. Contamination reaching monitoring well MW-4 at the northern site boundary had chemical signatures indicative of a spill near the processing building (CDPHE, 2016). There could be two sources of groundwater that may have facilitated the transport of the spill to well MW-4 at the site boundary. One, over the 10-year period from the start of operations to the breakthrough of the chloride pulse at monitoring well MW-4, periodic recharge supplying groundwater for intermittent lateral migration along the bedrock surface might be sufficient to move the contamination to well MW-4. Two, the volume of liquid in the spill was sufficient to drive the contaminants a lateral distance of 920 ft (280 m) to well MW-4. For the latter, it is presumed that such a large volume of a spill would have be recognized at the time of operations, thus indicating that the contribution of natural recharge to the contaminant arrival at well MW-4 could not be neglected. Analyses in CDPHE (2016) of chloride data indicate pulse duration of 22 months at MW-4 in 1988 and 1989, from which CDPHE estimated a time lag of 7.75 years with the assumption that the spill occurred at the beginning of leaching operations. The duration of the chloride pulse observed at the well corresponds approximately to the duration of leaching operations from 1977 to 1979. The chloride pulse provides quantification of flow and transport parameters along the base of the weathered bedrock and unconsolidated sediment groundwater system, which is discussed further in Sections 2.4, Groundwater Monitoring and in Section 6, Model and Transport Parameters.

Mancos Shale

The Mancos shale is widely known across the western U.S. as a thick [>2,000 ft (>610 m)], laterally extensive, and low permeability rock layer (Department of Energy, DOE, 2011). At large depths, natural fractures are tight and relatively impermeable due to continuous pathways in fractures being closed as a result of confining stresses. Cross-layer permeability (vertical in flat-lying bedded layers) is often much smaller than transverse permeability. CDPHE (2016), and references cited therein, rely on the reputation of the Mancos Shale as a relatively impermeable unit that would act as a barrier for contaminant releases reaching the designated uppermost aquifer. This section describes site-specific characteristics of the lower Mancos horizons, thicknesses across the site, and permeability estimates from field tests.

At the Durita site, the Mancos Shale is a thin, isolated remnant in a doubly-plunging syncline at the southern end of the collapsed salt basin referred to as the Paradox Basin. Durita is located in a small portion of the basin (Coke Oven Syncline) that is slightly offset from the trend of the remainder of the Paradox Basin that extends to the northwest. The axis of the Coke Oven Syncline crosses the northeast portion of the Durita site (Figure 2), which would suggest that the

groundwater flow in the northeast portion of the site shifts to a more northwesterly flow from the northerly flow for the rest of the Durita site.

Borehole logs from the monitoring wells indicate Mancos thickness ranges from 42 to 70 ft (13 to 21 m) at the Durita site. Estimates of thickness beneath the leach tanks, however, can be much smaller. As noted previously, the Mancos Shale at the Durita site is a lens that is constrained in extent by being wholly contained in a doubly-plunging syncline. At the edges of the lens, where the Mancos Shale pinches out, the Dakota Formation is the uppermost bedrock layer. The thickness of the Mancos Shale at the site varies from 0 to 80 ft (0 to 24 m) thick based on an isopack map of the area (Fox, 1982, Figure 11), excluding Mancos Hill. Based on visual estimates using Figure 11 of Fox (1982), thicknesses of the Mancos under the leach tanks may be described as:

- 0 to 60 ft (0 to 18 m) thick beneath leach tanks LT-201 and LT-202, with an average of <30 ft (<9 m)
- 0 ft thick under a portion of leach tank LT-202 at southwest corner of the site
- <20 ft (<6 m) for half the area of leach tank LT-202
- 30 to 80 ft (9 to 24 m) thick beneath LT-203, average of 60 ft (18 m)
- 40 to 80+ ft (12 to 24+ m) beneath the closure cell and evaporation ponds; descriptions of closure cell construction indicate that cell was built on "scarified bedrock," which implies some thickness of weathered Mancos Shale was removed.

For areas with thin or no Mancos Shale, the Mancos-Dakota transitional contact is the uppermost bedrock. The upper Dakota Formation is comprised predominantly of alternating shale and sandstone layers.

Beneath the Durita site, the Mancos Shale is a calcareous to carbonaceous gray shale with thin beds of iron-bearing sandstone (CDPHE, 2016; Fox, 1982). The Mancos grades into the underlying Dakota Sandstone, which contains some carbonaceous shale and coal but alternates between a shale and a friable to moderately cemented sandstone at shallow depths below the Durita site (Hecla, 1991). One or more apparently laterally extensive, thin sand layers near the bottom of the Mancos Shale were identified from boreholes as a conductive unit of the uppermost saturated layer. CDPHE (2016, p.8) suggests that the sandstone-claystone uppermost "water-bearing" unit appears to be 10 ft (3 m) thick. As more fully discussed in Section 3, this layer comprises the designated uppermost aquifer for compliance with the groundwater monitoring detection program. The interlayered shales and sandstones at the Dakota/Mancos transition and the thin sandstone layers in the lower Mancos appear to be hydraulically interconnected (CDPHE, 2016) and thus are combined as a single hydrogeological unit for the model inputs described in Section 6.2.

The extent of fracturing is not known at the site, but some fracturing is likely due to the site's location in the collapsed salt dome with its associated faulting in the Paradox Basin. Fractures found in bedrock at the surface may or may not be through-going; i.e., extending through the shale layer to the designated uppermost aquifer. Proving the absence of subsurface features like preferential pathways along fractures is difficult given the relative scale of the site, the scale of features of interest, investigative technologies, and flow times. The sparse number of

borehole logs and pumping tests are not adequate to conclusively negate the presence of through-going fractures. In addition, vertical boreholes inherently underrepresent near-vertical fracturing. Siting of wells can be difficult for finding conductive fractures. Hydraulically active fractures may be reflected by the perturbation of water levels during surface disturbing activities (NRC, 2007). However, the perturbation was a drop in water levels, which may also be explained by a change in the geomechanical conditions. Besides the structural imprint from the Paradox Basin evolution, near-surface processes may also affect the bedrock at the site. Exposures of the Mancos Shale near the surface generally exhibit fracturing and chemical alteration that produce a strong overprint of near-surface processes on the nature and character of the Mancos. The depth of influence for near-surface processes, the variable but thin thickness of Mancos, and the possibility and extent of through-going fractures leads to uncertainty in transport pathways at Durita. More complete discussions pertaining to the uncertainty of fracturing and the potential for flow pathways at the Durita site are contained in NRC 2014 and NRC 2007.

Variations in groundwater wells over time, other than during the 1993 reclamation period, in response to changes in precipitation rates were evaluated in NRC (2007) for indications of a direct hydraulic pathway through the Mancos, such as a through-going fracture. The groundwater monitoring program is described in Section 2.4, Groundwater Monitoring. Six of the seven wells monitored from 1991 to 1998 exhibited significantly confined conditions, and one well exhibited near unconfined conditions. The one well exhibiting near unconfined conditions is also the well exhibiting the widest range in chemical conditions (see Geochemical Conditions subsection of Section 2.4), which may reflect direct recharge. Monitored groundwater levels of the seven wells between 1991 and 1998 exhibit a potentially dampened and delayed response to changes in precipitation (NRC, 2007). The small variations in the responses between wells could be explained either by variations in direct recharge between the wells, or by recharge upgradient in the watershed with variations in the conductive units between wells. Based on borehole descriptions, the conductive horizons at each of the wells are not prominent identified and characteristics at each well may not be similar. They are described variably as a single to multiple thin horizons. Also, with some exceptions, the groundwater water levels in the seven wells act in concert, which would not be expected if specific wells encountered through-going fast pathways. However, seven wells in the 160 acres site area could readily miss areas of groundwater levels responding directly to recharge through potential vertical fractures.

Laboratory and field measurements of permeability were performed on the Mancos Formation at the site. Laboratory measurements on core samples reflect only the matrix permeability of the hydrogeologic unit. Therefore, the focus is on field measurements that would better reflect the permeability at a scale appropriate for input to model parameters and possibly reflect the influence of preferential pathways such as lateral flow along fractures or bedding planes. Borehole percolation tests at set horizons were performed at locations within and near the site. Field percolation tests for hydraulic conductivity in the upper 15 feet (4.6 m) of unconsolidated sediments and weathered Mancos at 20 boreholes (Fox, 1977, 1982) averaged 15 m/yr (48 ft/yr) with a maximum of 103 m/yr (340 ft/yr). Field percolation tests for hydraulic conductivity at

17 locations in the unweathered Mancos horizons lower in the column (Fox, 1982) averaged 11 m/yr (37 ft/yr) with a maximum of 98 m/yr (320 ft/yr).

In summary, the uncertainties described here and in NRC (2007 and 2014) include the (i) possibility of fracturing in this structural environment with unknown spacing, (ii) assessment of hydraulic conditions at the site, and (iii) potential that the monitoring well network did not capture the possibility of fast flow and transport pathways. To address these uncertainties, a hydraulic connection of the ground surface and saturated horizons is incorporated into the consequence modeling later in this report by conservatively assuming through-going fractures between the leach tanks and the designated uppermost aquifer.

Aquifers Below the Mancos Shale

Fox (1982) identified four aquifers in the Coke Oven Basin and provided approximate depths: (i) basal unit of Dakota Sandstone at 225 ft (69 m), (ii) Burro Formation at 290 ft (88 m), (iii) Burro Formation at 355 ft (109 m), and (iv) Brushy Basin Member of the Morrison Formation at 440 ft (134 m). In the same study, Fox (1982) indicated that several sandstone and shale beds of the basal Mancos and upper Dakota Sandstone were "partially to nearly saturated," but did not label these as aquifers due to low yield and limited spatial extent.

All four aquifers below the Mancos Shale were described by Fox (1982) as confined. At two deep wells in the study, the potentiometric surface rose to depths of 22 and 35 ft (6.7 and 11 m) below the ground surface. Fox (1982) states that eight different bentonite/clay layers occur between the ground surface and the aquifer in the lower Dakota Sandstone; bentonite layers are generally considered to be proficient aquitards separating aquifers. In addition, the groundwater composition changes from calcium-sulfate above 200-ft (61-m) depths to calcium-bicarbonate composition below. The two upper aquifers were described as brackish, and the two lower ones as fresh water. The well at the Coke Oven Ranch north of Durita (Figure 1) taps the basal unit of the Dakota Sandstone and produced 30 gal/min (164 m^3/d) upon completion of drilling (Fox, 1982). Support for the conclusion in Fox (1982) that there is not a significant hydraulic connection between the lower aquifers and the saturated layers in the Mancos Shale in the Coke Oven Basin includes the (i) large magnitude of confined condition of the aquifers in the Dakota Sandstone and Burro and Morrison Formations, (ii) chemical stratification of the aquifers, and (iii) the multiple bentonite horizons that occur along with shale layers.

2.3   Site Engineered Features

The site contains four disposal areas (three leach tanks and one closure cell), which together comprise a tailings management system (term used by CDHPE). For the modeling in this report, the important aspects of the tailings management system are the cover and liner performance and the distances to the property boundary. The property boundary is approximately the fence line on the northern edge beyond the location of decommissioned evaporation ponds. The important distances for the modeling, such as from the leach tanks to the site boundary are described in Section 6, Modeling Input.

Compacted clay layers were emplaced at Durita as liners for the leach tanks, evaporation ponds, and closure cell, and as a component of the covers for the leach tanks and closure cell. Besides a 1-ft thick compacted clay layer, the covers include a 5.3- to 7.5-foot (1.6- to 2.3-m) thickness of soil that acts both to protect the compacted clay layer from surface processes and to reduce radon emanation at the ground surface (CDPHE, 2016). The material for the compacted clay layers was derived from natural clay-rich unconsolidated sediments from the local area that were compacted to 1-ft (30-cm) thicknesses following a compaction protocol to 85 percent of the laboratory-based Proctor test (American Standard Testing Method ASTM-698). Compacting to the designated bulk density to the wet side of the line of optimums from the Proctor test[1] generally ensures that the field installed compacted clay layer is devoid of interconnected macropores (Benson, 2001) that may lead to higher hydraulic conductivity values. The design criteria for the compacted clay layers for flow of water is a hydraulic conductivity of $1e10^{-7}$ cm/s or 0.032 m/yr (0.1 ft/yr), which was demonstrated for Durita by laboratory core sample measurements that fell below the design hydraulic conductivity value (CDPHE, 2016).

A survey of the cover and liner literature was undertaken on the long-term performance of cover and liner systems that included a compacted clay layer. Information on performance of compacted clay layers for uranium tailings is sparse, but performance testing data are more widely available for compacted clay layers in other applications. The compacted clay layers in other applications (e.g., landfills) are generally thicker [up to 3 ft thick (1 m)] and often constructed using interleaving compressed lifts. Hence, the Durita compacted clay layers would expect similar or lesser performance compared to compacted clay layers in other applications. The environments and evolution of compacted clay layers in liners are different from those for covers, and thus are described separately below.

Performance of compacted clay layers in covers has been assessed using field observations and long-term field tests (Benson et al., 2011). Hydrologically, the clay layers in covers gradually evolve over periods of months to years due to surficial processes. Surficial processes include plant roots, burrows, temperature and moisture cycling, and distortion (Benson et al., 2011). Based on observations relevant for long-term performance of a compacted clay layer below a soil layer in field tests, a summary of field test results (NAS, 2007) showed a five-fold increase in flux through the cover systems from initial results to long-term results completed 7 years later. Benson et al. (2011) concluded that covers with compacted clay layers evolve to become similar hydrologically to the natural state of the unconsolidated sediments in the area. Thus, the water flux through the covers evolves towards the natural recharge rate for soils of the area. Therefore, in the modeling described in this report for a 1,000-yr performance period, the covers are assumed to be a barrier to flow only to the extent that the natural unconsolidated sediments at Durita are a barrier.

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[1] A Proctor test determines the relationship between initial dry density and moisture content. Compacting soils to a density and moisture that plots above the line of optimums produces a low permeability liner based on the presumption that the soil is wet enough such that the micropore are saturated and the macropores will be eliminated during compaction.

Performance of compacted clay liners have been summarized in Benson et al. (1999), Environmental Protection Agency (EPA, 2002), and Moo-Young et al. (2003). Liners, because of their depth, would not be as readily affected by surficial processes and thus may remain more stable for longer periods. However, recognizing that field performance of compacted clay liners differs from results based on laboratory core measurements, the summaries in the literature focused on field measurement of flux through the barriers and field measurement of hydraulic conductivity. The field measurements include long-term test sites and actual impoundments that use compacted clay liners. The field test sites generally had compacted clay liners that were 2 to 3 ft (60 to 90 cm) thick, as compared to the 1-ft (30-cm) thick liner at Durita. As one would expect, EPA (2002) found that hydraulic conductivity decreased with an increase in compacted clay liner thickness. Moo-Young et al. (2004) and EPA (2002) report that 25 to 30 percent of the field-based hydraulic conductivity measurement results did not meet the design criteria of 1e-7 cm/s or 0.032 m/yr (0.1 ft/yr), even though associated laboratory-based core measurements met or easily exceeded the 1e-7 cm/s or 0.032 m/yr (0.1 ft/yr) design criteria.

Reductions in design performance of compacted clay liners are widely accepted to be primarily attributable to mechanical perturbations, evaporation during installation, and corrosive liquids after installation (NAS, 2007; LaTouche and Garrick, 2012). Mechanical effects include compaction of overlying load, puncturing, and differential settling. Evaporation from exposure of compacted clay layers to the atmosphere, even for short periods during installation, was seen to lead to drying and subsequent reductions in performance due to clay shrinkage and the formation of open cracks. Chemical incompatibility of the pore water and compacted clay liner material can lead to degradation of the compacted clay liner performance. However, such an assessment is highly complex due to the processes involved (e.g., adsorption, ion exchange, and precipitation) and the range of leachate chemistry and mineralogy of the systems found at different sites. Simplifying chemical incompatibility to an assessment of the effects of pH, solutions coming into contact with compacted clay liners have not been shown to be detrimental to the integrity of the liner unless the pH of the solution is pH<2 or pH>13 (Benson 2001). Unlike observations of defects per acre for geomembrane liners, there are no similar data for compacted clay liners in the literature – the type and cause of defects would be different for the two types of liners.

At Durita, processes that lead to reduction in compacted clay liner performance may include mechanical perturbations, evaporation, and chemical incompatibility. For mechanical perturbations, the impoundments at Durita have not yet shown signs of differential settling that could affect the liner (CDPHE, 2016, p.55). However, there is no description in available Durita documents of the installation procedure that might reflect on the potential for initial defects. Additionally, the Environmental Report (Four Corners ER, 1977) described the plan for loading the leach tanks by dumping the agglomerate on the berm and pushing piles into the leach tank using a bulldozer. Besides the potential mechanical disturbances of the compacted clay liner during loading, this plan would leave a period of time for portions of the compacted clay liner being exposed, and thus, a potential for significant evaporation to occur with associated drying and crack formation. For chemical incompatibility at Durita, the sulfuric solutions introduced into the leach tanks had extremely low pH. Based on the measurements from the effluent in the

raffinate and evaporation ponds, the pH of the solution phase was as low as pH=1.85 (AKGeoconsult, 1993). Thus, the potential exists for geochemical alteration of the compacted clay liner to the extent that hydrological properties may have been adversely modified from its original design specifications. However, laboratory tests using 5% sulfuric acid for Durita indicated that the hydraulic conductivity of a compacted clay remained within the range of the original compacted clay (CDPHE, 2016, p.14).

This discussion of potential degradation of compacted clay liner performance is based on observations at other sites. Other than the localized leakage at the evaporation ponds described in Section 2.4 Groundwater Monitoring, there are no other indications of potential problems with the compacted clay liners at Durita. Uncertainties in the permeability of the compacted clay liners and related flux through the barriers are addressed by the conservative selection of parameter values in Section 6 and sensitivity analysis of Sections 7.

2.4   Groundwater Monitoring

Two sets of monitoring wells were developed on the site with sequential (non-overlapping) periods of operation. Both sets are shown on Figure 2. CDPHE (2016) stated that both sets were considered part of the detection monitoring program.

The first set, MW-2 to MW-7, was installed in 1977 and monitoring was discontinued in 1991. CDPHE (2016, Section 7.4.2.2) noted, however, that monitoring data records were lost for the period 3rd quarter of 1981 to 2nd quarter of 1985. A significant use for the MW-2 to MW-7 monitoring data was to provide an indication of pre-operation period groundwater chemistry, such as was utilized by Studious Solutions (1994) for analysis of background concentrations. The wells were plugged in 1991 because of potential well construction concerns. CDPHE (2016, p.23) suggested that the wells monitored the unconfined upper water-bearing zone, though a review of the records for the MW-2 to MW-7 wells indicated that the construction and completion techniques used for the wells could allow surface water to enter the wells (CDPHE (2016, p.23). CDPHE, however, indicated that no conclusive evidence was found of surface infiltration into the wellbore. According to the Four Corners ER (1977), the wells were sealed with clay around the collar at the ground surface to eliminate surface inflow. Indicative of a different issue with the wells, the Four Corners ER (1977, p.44) stated that the MW-2 through MW-7 wells were purposely left open from the upper layers to the well bottom so that measurements reflect integrated groundwater conditions encompassing all zones. Thus, the horizon (unweathered Mancos or unconsolidated sediment layer) controlling the water levels in the wells or the source of contamination would be ambiguous.

A second set of monitoring wells (MW-8 to MW-14) were installed in 1991 with screened intervals set to capture only the conditions in the compliance-designated uppermost saturated zone at the Mancos/Dakota transition and the thin sandstone layers near the base of the Mancos Shale. None of the new wells were open to the unconsolidated sediments overlying the Mancos Shale. Quarterly monitoring in the new wells began in the third quarter of 1991, was stopped in 1998, and the wells were plugged in 2002. CDPHE (2005c) listed MW-11 and

MW-12 as compliance wells and MW-8 and MW-14 as background wells. The other three wells are cross-gradient wells, meaning they are located off to the side of a transport pathway. Monitoring stopped with CDPHE concurrence well before submittal of the final CRR (CDPHE, 2016) and before termination of the license, which is the required period of time for ending the monitoring program under 10 CFR Part 40.

The locations of the compliance wells are not appropriate for a detection monitoring program, especially considering the expected transport times at the site and the distance from the source to the designated compliance wells. All the wells operated from 1991 to 1998 (MW-8 to MW-14) are located near site boundaries. The primary compliance well for the leach tanks, MW-11, is located approximately 500 m (1,640 ft) in the main transport path downstream from the potential source, while wells MW-9, MW-10, and MW-13 are slightly closer but would be on the lateral edge of transport pathways from the leach tanks. Considering the highest measured permeability and groundwater gradient values for the designated uppermost aquifer, a release occurring at the start of operations in 1978 would only be detected in a well that was placed within 65 m (213 ft) of the leach tank (excludes dispersion) at the time the monitoring program was stopped in 1998. Therefore, acknowledging the practical aspects of locating wells for a detection monitoring at Durita, the NRC staff has taken a practical approach and reconciles the inadequacy of the detection monitoring program through the use of conservative scenario modeling of a release in this report to reach part of its decision on whether to concur or not on the CRR.

Monitoring of Groundwater Contaminants

Hecla-Durita 1993 License Condition No. 26 from the State of Colorado required quarterly groundwater monitoring of total dissolved solids, chloride, sodium, carbonate, bicarbonate, sulfate, arsenic, molybdenum, selenium, gross alpha, gross beta, "Ra-225," thorium-230, and uranium (CDPHE, 2016, page 21; radium-226 was likely intended instead of Ra-225). CDPHE (2016) indicated that these constituents were detected in the groundwater or were found in the tailings waste material. A comparison of upgradient and downgradient wells was performed for six elements commonly found at similar type milling operations. They indicated that none of six elements occurred at concentrations or frequencies that would indicate a release (Studious Solutions, 1994). Of the six elements used in the statistical analysis, molybdenum was not found in the Durita tailings, and radium and thorium have high distribution coefficients and would migrate slowly. The remaining three elements, selenium, arsenic, and uranium, were selected as indicator elements (CDPHE, 2016, p.15). CDPHE (2016, page 8) concluded from the analysis of the data from the groundwater monitoring program that no releases from the leach tanks or closure cell have reached the designated uppermost saturated zone. CDPHE based their conclusion on statistical analyses of upgradient and downgradient wells. However, four events or observations reflected in the monitoring records warrant discussion here.

A 22-month spike in chloride values at well MW-4 began in 1988 and ended in 1989. CDPHE (2016, page 89) suggested that a leak from the area near the processing and packaging building (Plant Area in Figure 2) during operations (1977-1979). Sodium and ammonium

chloride were used in the processing steps at those buildings. Based on borehole logs in the Four Corners ER (1977), MW-4 is one of the two early wells where the screened portion of the well may have included the weathered Mancos, which would be hydraulically connected to groundwater in the unconsolidated sediments. Both MW-4 and MW-5 are located in the main wash (arroyo) described in the Four Corners ER (1977, page 22) as being a possible zone of shallow groundwater in the unconsolidated sediments. CDPHE (2016) noted that, as with other wells in the early set of monitoring wells that operated 1978 to 1991, MW-4 may not have been properly sealed near the ground surface, which implies an overland flow pathway for the chloride. A description of the early well (MW-2 to MW-7) construction in the Four Corners ER (1977) indicated that they were sealed with clay packed around the collar at the ground surface (Four Corners ER, 1977). Considering that overland flow of a leak in the Plant Area would have been readily observable and unlikely hydrologically given the permeable unconsolidated sediments, the most likely pathway is along the base of the weathered bedrock and unconsolidated sediments. This event is noteworthy because estimates of transport parameters for a pathway in the unconsolidated sediments at the bedrock interface may be derived from the timing of the pulse and distance to MW-4. Based on the decrease in chloride after the pulse, which would reflect the end of a leak, this release is not likely a continuing concern for evaluating future contamination migrating offsite. Rather, this event provides information on transport properties for one possible pathway for migration of contaminants from the leach tanks situated immediately upgradient of the Plant Area, and is discussed further in Section 6, Model and Transport Parameters.

Elevated uranium results were found in wells MW-2 to MW-7 in two different quarters in 1979, though not all in the same sampling round (CDPHE, 2016, page 91). In addition, elevated radium-226 and lead-210 results were found in 1978 and 1980 in both upgradient and downgradient wells relative to the leach tanks. CDPHE (2016) stated that distribution and timing of elevated values is typical of laboratory or analytical sample contamination. In the 1980 case, the values for dissolved solids did not reflect a corresponding increase, which would support the laboratory artifact conclusion for that particular instance.

Elevated uranium concentrations were initially found in five of the seven new wells in 1991, but concentrations decreased thereafter (CDPHE, 2016, page 18; Hecla, 1998a). Also, all wells exhibited spikes in radium-226 in the first half of 1993, except MW-14. CDPHE (2016) suggests that some increased radionuclide concentrations may be due to laboratory errors. In addition, comparative analyses of upgradient and downgradient wells indicated that the statistical difference between upgradient and downgradient wells was caused by the number of detections in wells upgradient of the leach tanks being greater than the number in downgradient wells (Hecla, 1998).

The NRC staff assessed the CDPHE conclusions by considering additional observations and information. Perturbations of the groundwater system, such as seen in the 1993 groundwater levels (NRC, 2007) may be related to the 1993 spikes in radionuclide values. Reclamation activities were underway in 1993. Two of the wells with the highest concentrations of uranium are upgradient or cross-gradient to the leach tanks. Coincidentally these two wells, MW-13 and

MW-14, are adjacent to access roads. These two wells also exhibit the highest alkalinity values of all the wells operated from 1991 to 1998 (MW-8 to MW-14), which may be significant because the Naturita groundwater contamination showed a strong positive relationship between uranium in solution and alkalinity (Davis and Curtis, 2003). The Mancos Shale itself contains significant quantities of uranium, which depending upon a change to more oxidizing conditions, could be released into the groundwater. Uranium values found in the groundwater at Durita fall within the range reported by DOE (2011) for groundwater associated with the Mancos Shale across the region. In summary, the NRC staff finds that the radionuclides in the groundwater may not be related to possible releases from the leach tanks. Rather, the radionuclides may be derived from the Mancos Shale with local and temporal variations in the data from the monitoring wells explained by changing conditions, incidental contamination of the wells from the ground surface, or as suggested by CDPHE (2016), laboratory artifacts.

Lastly, isolated contamination was found below the liner of the evaporation ponds at the site (CDPHE, 2007, Gibb's memorandum attachment). The evaporation ponds remained in use from the time of operations (1978-1979) until reclamation in 1994 (CDPHE, 2016, page 47). Localized leaks occurred through the liner of the evaporation ponds as identified by small zones of contamination found immediately below the liner. Excavation of those zones continued until gamma scans indicated the contaminated material was removed. Where the evaporation pond liner was located directly on bedrock, the contamination was limited to depths of an "inch or so" (CDPHE, 2007) below the liner. Where the liner was emplaced on unconsolidated sediments, the contamination was limited to a "few feet" laterally or vertically. After removal of all hotspots beneath evaporation ponds, the contaminated material was placed in the closure cell.

CDPHE (2016) concluded from the analysis of the data from the groundwater monitoring program that no releases from the leach tanks or closure cell have reached the monitoring wells in the designated uppermost saturated zone. The CDPHE based their conclusion on observations from the two sets of wells that together covered the period beginning during operations in 1978 and ending in 1998, including the use of statistical analyses of upgradient and downgradient wells. The NRC staff evaluated events in the monitoring records associated with elevated contaminant levels and agrees with CDPHE's assessment that evidence of releases from the leach tanks are not reflected in the monitoring well data. However, since no well data were collected in the last 22 years (i.e., since 1998), and because the wells were not adequately placed to capture releases associated with potential isolated fractures or sufficiently close to the source given expected transport times based on measured hydrologic information (e.g., permeability and gradients), models for transport scenario pathways are developed in Sections 4 and 5 in this report to evaluate the consequence of hypothetical releases from the leach tanks to inform the NRC's overall assessment of the ability of the Durita site to control radiological hazards over the performance period in 10 CFR Part 40, Appendix A.

Observations on Geochemical Conditions

The spatial distribution of total dissolved solids and alkalinity may reflect on the conceptualization of groundwater flow and transport near the site. The first NRC staff

observation is that the variation in total dissolved solids cannot be readily linked to contributing source of the groundwater, whether it be the Mancos groundwater, unconsolidated sediment groundwater, or surface water leakage into the well. Total dissolved solids ranges from 1,600 to 6,000 mg/L for all the wells (Hecla, 1998a), which dominantly reflects spatial variation. Distinct individual ranges reflecting temporal variations for each well are much smaller based on data in Studious Solutions (1994, Tables C and 1 to 7).

- The temporal variation at individual wells can be divided into two groups: wells with little variation (total dissolved solids range <300 mg/L) and those with total dissolved solids ranges that span >1,000 (up to 2,400 mg/L span). Of the wells sealed at the top and screened only to the lower Mancos Shale, MW-12 is the only well that reflects unconfined or slightly confined conditions based on water levels, and has the widest temporal range of total dissolved solids values.
- Three of the four wells with the lowest mean total dissolved solids concentration are the three upgradient wells (MW-5, MW-8, and MW-14), and the fourth well (MW-13) is a cross-gradient well on the east boundary of the property.
- Two of the three highest mean total dissolved solids concentrations are wells near the Plant Area (Figure 2) on the west side of the site, and the third well (MW-6) of the three is in the east central side of the site near other operations.

The idea that direct vertical recharge may be the cause of the wide temporal variation in total dissolved solids values is not supported by data from the six early wells (MW-2 to MW-7) that exhibit little temporal variation. Note that these early wells may reflect water from the Mancos Shale, unconsolidated sediment layer, or potentially the surface. The later wells (MW-8 to MW-14) were screened only in the lower Mancos. The spatial variation in total dissolved solids, which is positively correlated with sulfate concentration, may be explained in two ways. Either local groundwater conditions are significantly affected by the local mineralogical control on geochemical conditions, or groundwater contributions are derived from different sources such as upgradient recharge versus direct vertical recharge.

The second NRC staff observation concerns alkalinity, which affects transport characteristics of uranium at the site. The importance of alkalinity is illustrated by uranium migration at the nearby Naturita site that was observed to be inversely correlated to alkalinity levels (Davis and Curtis, 2003). Most of the wells at Durita exhibit alkalinity levels in the lower end of the range at Naturita. All of the wells in the area of the primary transport pathway from the leach tanks (see Section 4) exhibit low alkalinity. Two wells on the eastern edge of the site, MW-12 and MW-14, exhibit high alkalinity. At Naturita, these higher alkalinity levels correspond to areas with high uranium groundwater concentrations and low estimates of distribution coefficient values. These two wells on the east side of the Durita site are not associated with the primary transport path considered in Section 4. At Durita, the primary expected transport pathway for the bulk of the tailings falls in the areas of low alkalinity represented by all the other wells at the site, thus suggesting the use of a higher value for the uranium distribution coefficient. Ranges for the uranium distribution coefficient at Naturita and for a wider variety of sites is discussed in detail in Section 5, Source Term and Geochemical Behavior.

3 Designated Uppermost Aquifer

This section discusses the point of compliance, the designated uppermost aquifer, the groundwater quality criteria for the designated uppermost aquifer, and the chemical benchmarks to use as reference points for presenting modeling results in the remainder of the report. Because no contamination was found in the detection monitoring program, benchmarks based on drinking water standards are used to assess consequences of a release in the modeling exercise even though background values at the did not always meeting drinking water standards.

3.1 Assessment of Aquifers

This section describes and assesses the aquifers and saturated horizons in the vicinity of the Durita site in terms of the regulatory requirements. Point of compliance is the location in the designated uppermost aquifer where the groundwater protection criteria must be met (10 CFR Part 40, Appendix A). The uppermost aquifer refers to "the geologic formation nearest the natural ground surface that is an aquifer, as well as lower aquifers that are hydraulically interconnected with this aquifer within the facility's property boundary" (10 CFR 40). Implicit in the regulation is that protection of the uppermost aquifer also protects lower aquifers. Aquifers lower in groundwater system would also be protected if it can be shown that there is not a hydraulic connection with the uppermost aquifer. A factor in the discussion in this section is the definition of aquifer. From 10 CFR Part 40, Appendix A, "aquifer means a geologic formation, group of formations, or part of a formation capable of yielding a significant amount of groundwater to wells or springs."

Description of Aquifers and Saturated Horizons

The uppermost aquifer designated by CDPHE for monitoring is one to several thin saturated sandstone layers at the base of the Mancos Shale and the transition to the Dakota Formation (CDPHE, 2016). Alternating layers of shale and sandstone mark the boundary between the Mancos Shale and Dakota Sandstone. Fox (1982) indicated that several sandstone and shale beds of the basal Mancos and upper Dakota Sandstone were "partially to nearly saturated," but did not label these as aquifers due to low yield and limited spatial extent. Pump rates from the onsite monitoring wells were reported to be 1 gal/min (5.5 m^3/d) or less (Hecla, 1996, 1998a; Studious Solutions, 1994). For comparison, a lower threshold for a single-family residential well is 6 gal/min (33 m^3/day). Paleo-erosion of the syncline in the valley exposed the lower Mancos at and immediately south of the site, which limits the spatial extent of the saturated horizon thus constraining any sustained aquifer production. The lower Mancos horizon is exposed at the ground surface at the southwestern corner of the site.

Overlying the unweathered Mancos Shale at the Durita site is an unconfined groundwater system comprised of unconsolidated sediments and weathered bedrock. Water from precipitation and runoff sporadically infiltrate unconsolidated sediments, and could either migrate along the interface with the bedrock or flow vertically into the bedrock if conductive

fractures are present. At least some of the infiltrated water migrates along the bedrock contact as indicated by (i) the chloride pulse likely derived from a release at the processing building and reaching MW-4 near the site boundary, and (ii) altered rock in the bedrock-unconsolidated interface in exposures at Dry Creek. For the latter, the alteration may reflect conditions during a past, wetter climate. The groundwater in the unconsolidated sediments is generally unsaturated, though may intermittently be saturated in channels and pockets at the bedrock surface (see Section 2.2). Fox (1982) suggested that the water quality of this horizon was similar to that of the surface water in Dry Creek. Reporting on previous investigations, Fox (1982, Table 5) indicated the sulfate and total dissolved solid levels in the alluvial aquifer were significantly above the EPA's secondary drinking water standards. There is no current user of the surface water in the ephemeral Dry Creek near Durita, nor downstream to the San Miguel River. Dry Creek breaches the otherwise closed basin, entering from the west and exiting to the east. Fox (1982) reports the existence of upstream diversion of surface water in Dry Creek for irrigation to the west of the basin.

Underlying the designated uppermost saturated layer in the Mancos Shale is the Dakota Sandstone, the latter of which contains several saturated horizons. Fox (1982) identified four aquifers in the Coke Oven Basin in order of increasing depth: (i) basal unit of Dakota Sandstone at 225 ft (69 m), (ii) Burro Formation at 290 ft (88 m), (iii) Burro Formation at 355 ft (109 m), and (iv) Brushy Basin Member of the Morrison Formation at 440 ft (134 m). Shale layers, including several thin bentonite layers, occur in the middle of the Dakota Sandstone unit and in the transition to the Mancos Shale (Fox, 1982).

The only known producing well in the basin is located at the Coke Oven Ranch on the northeast side of the Coke Oven Syncline. The residence and well are approximately 0.5 miles (0.8 km) north of the Durita site (Figure 1); no other residences are within several miles of the site (DOE, 2005). The well at the Coke Oven Ranch likely reaches the lower portion of the Dakota Sandstone (Fox, 1982) where the water quality is sufficient for residential use. The Durita site and Coke Oven Ranch are on opposite sides of the syncline. As a closed basin, flow in the saturated layers following bedding planes of the Dakota Sandstone is controlled by synclinal geologic structures, except as interrupted by faults. The groundwater in the Dakota Sandstone below the Coke Oven Ranch likely flows southwesterly towards the middle of the doubly-plunging syncline. Groundwater below the Durita site flows northerly to northwesterly from the opposite side of the syncline towards the middle of the doubly-plunging syncline. As the water tapped by the Coke Oven Ranch well is not in the flow path from the Durita site, any release from the Durita site would not pose a threat to the existing well.

Hydraulic Connection with Lower Aquifers

There are several observations at the site that reflect on the hydraulic connection between the designated uppermost aquifer and aquifers lower in the Dakota Formation that are pumped for residential purposes in the valley. Staff notes that well placement in fractured terrains is important when assessing zones of higher production and ascertaining hydraulic connection. There is not enough characterization data available on the site, however, to determine the

possible presence of fracture capable of carrying flow (NRC, 2007; NRC, 2014). However, available water levels and permeability values at 13 wells (CDPHE, 2016) on the 160-acre (0.65 km^2) site and 17 boreholes (Fox, 1982) within and near the site do not reflect hydrologic characteristics indicative of fracture flow, which suggests that a high density fracture network does not exist. A more widely spaced fracture network could readily be missed by the wells and boreholes, which is an uncertainty that is covered in the transport scenario pathway assessment in Section 4.

There is information, however, that reflects on the hydraulic connection between the designated uppermost aquifer in the Mancos Shale and aquifers in the lower portion of the Dakota Formation, which is below the Mancos Shale. All four aquifers below the Mancos Shale were described by Fox (1982) as confined. At two deep wells in the study, the potentiometric surface rose to depths of 22 and 35 ft (6.7 and 11 m) below the ground surface. Fox (1982) states that eight different bentonite/clay layers occur between the ground surface and the aquifer in the lower Dakota Sandstone; bentonite layers are generally considered to be proficient aquitards separating aquifers. In addition, the groundwater composition changes from calcium-sulfate above 200-ft (61-m) depths to calcium-bicarbonate composition below. The two upper aquifers were described as brackish, and the two lower ones as fresh water. The relationship that brackish groundwater overlies fresh water is inconsistent with the general trend of increasing total dissolved solids with depth related to infiltration of chemically dilute precipitation. In addition, the well at the Coke Oven Ranch taps the basal unit of the Dakota Sandstone and produced 30 gal/min (164 m^3/d) upon completion of drilling (Fox, 1982). The pump rates from the onsite monitoring wells were reported to be 1 gal/min (5.5 m^3/d) or less (Hecla, 1996, 1998a; Studious Solutions, 1994), which suggests that the onsite wells were not hydraulically connected to the lower aquifers where the production capability is much higher. CDPHE (2007, attachment) reported that the onsite "production well" produced at 11 gal/min (60 m^3/d), although the depth of well was not reported. This well was not sufficient for site water use, which instead relied on trucking water to the site for operations.

The staff concludes that there is not a significant hydraulic connection between the saturated layers in the Mancos Shale and aquifers of the lower Dakota Formation used for water supply in the Coke Oven Basin because the (i) large magnitude of confined condition of the aquifers in the Dakota Sandstone and Burro and Morrison Formations, (ii) chemical stratification of the aquifers, and (iii) the multiple bentonite horizons that occur along with shale layers.

Definition of Uppermost Aquifer

The CRR designates the uppermost aquifer for compliance to be the lower portion of the Mancos Shale. In the description of this horizon, Hecla (1992) indicated that it is not a potential water resource because (i) the quantity and yield rate are too low, (ii) the areal extent is limited, and (iii) the water quality is poor. Based on site data, the NRC staff agrees with this characterization of the saturated horizons near the base of the Mancos Shale at Durita. While this characterization raises questions as to the application of the definition of uppermost aquifer in NRC regulations, especially as capable of "yielding a significant amount of groundwater to

wells," designation of this horizon is conservative in that it is not "interconnected" (hydraulically connected) with natural aquifers lower in the system, as described in Section 3.1. CDPHE (2016) designated this horizon as the uppermost aquifer, the NRC staff uses the term "designated uppermost saturated layer," but the NRC staff finds the CDPHE designation to be conservative and therefore acceptable for its analysis of safety.

Summary

The CDPHE designated uppermost aquifer is not in hydraulic connection with aquifers lower in system that are used as a source of water for local residents. Furthermore, while the CDPHE designated uppermost aquifer in the sandstone layers of the lower Mancos does not clearly fit the regulatory definition in 10 CFR Part 40, this designation is conservative in terms of public safety, and is therefore appropriate for use in this groundwater evaluation. Therefore, in the consequence modeling of this report, this horizon is assumed to be the uppermost aquifer and modeling results are presented accordingly.

3.2   Point of Compliance Used in Modeling

The north side of the site is marked by a fence line at the boundary and is 494 m (1,620 ft) downstream of the leach tanks and 94 m (308 ft) downstream of the closure cell. These lengths are measured from the most downstream edges of the impoundments. The hypothetical well in this model is placed at the site boundary in Section 6. Note that the terminology of point of compliance wells, point of exposure wells, and associated safety analysis is not used here because those terms imply detected contamination, which is not the case at Durita.

The presence of the well at the Coke Oven Ranch (a single-family dwelling), and the use of the groundwater for cattle and horse grazing, is indicative of potential for future groundwater use anywhere in the basin. Ignoring the poor groundwater quality and low production capability of the designated uppermost saturated layer, the modeling in Sections 4 to 6 considers the potential in the future for a well to be located immediately downgradient of the Durita site boundary where contamination could reach the designated uppermost aquifer.

3.3   Benchmarks for Groundwater Constituents

A conservative approach is taken to use drinking water standards when available as benchmarks to assess consequence modeling results. Additional conservatism occurs as a result of the designated uppermost aquifer's limited production Use of drinking water standards is conservative because compliance criteria such as groundwater protection standards following 10 CFR 40, Appendix A, Criterion 5B(3) and 5B(6) consider water use scenarios, and consequently may be higher than drinking water standards. Criterion 5C criteria for radium-226 is included in Table 1 and is used as a benchmark in the assessment of modeling results. A gross alpha value of 15 pCi/L is also listed in 10 CFR, Appendix A, Criterion 5C. The discussion below provides the rationale for selecting a subset of the constituents of concern identified in the CRR for inclusion in the source term for RESRAD modeling in Sections 6 and 7.

The CRR list of the constituents of concern includes total dissolved solids, chloride, sodium, carbonate, bicarbonate, sulfate, arsenic, molybdenum, selenium, gross alpha, gross beta, radium-226, thorium-230, and uranium (CDPHE, 2016). The EPA Drinking Water Standards and Durita background water quality provide a context for simulation results. Table 1 provides the EPA criteria and the range for selected radionuclides measured at the groundwater monitoring wells based on data from Fox (1982), Hecla (1998a), and including summaries in CDPHE (2016).

Uranium-238 and radium-226 will provide the main comparisons with simulation results in Section 7 because uranium and radium are the most mobile elements of the radionuclides considered. Simulations in Sections 7 report results in units of radioisotope activity of picocuries per liter (pCi/L) – not dose results as typically reported from RESRAD simulations. Conversion of simulation results reported in activity concentration is required for comparison with the criteria for lead and the total uranium toxicity criteria values in Table 1, which are in units of mass-based concentration for toxicity. A uranium-238 activity concentration of 10.2 pCi/L (377 Bq/m^3) is comparable to 30 μg/L total uranium, and a lead-210 activity concentration of 0.0011 Ci/L (4.2e10 Bq/m^3) is comparable to 0.015 mg/L lead. The contrast between the two radionuclides is due to the eight orders of magnitude difference in their respective specific activity constants, 3.4e-7 and 76 Ci/g, respectively. Also, uranium-234 can be ignored in the total uranium calculation because it does not contribute meaningfully to the mass-based concentration. Lead toxicity will not be a concern at Durita because the criteria in Table 1 are several magnitudes larger than the peak lead-210 values produced as part of the uranium decay chain in the modeling described in Section 7, Modeling Results. Thorium has no applicable groundwater protection standard, but would be encompassed in the gross alpha standard along with the other significant alpha emitters uranium-238, uranium-234, radium-226, and polonium-210. As is discussed in Section 5, thorium-230 is not likely to migrate offsite, but may occur offsite due to ingrowth from uranium that has migrated. Gross alpha is not considered here for a benchmark in the modeling because the primary alpha emitters that would migrate offsite are directly considered as individual benchmarks. Vanadium is not included in the EPA Primary or Secondary Drinking Water Standards. But the EPA posted[2] the State of Indiana criteria for the toxicity of vanadium which recommends values of 2300 μg/L and 230 μg/L for nonhuman and human ingestion, respectively. The State of California set a notification level of 50 μg/L for vanadium in drinking water (Wright et al., 2014), which is the value listed in Table 1.

The Durita background water quality was described as poor in Section 2 because of the high sulfate and total dissolved solids. These constituents are not part of the simulations in Sections 6 and 7, but are mentioned here because of the EPA Secondary Drinking Water Standards. Total dissolved solids levels at Durita range from 1,600 to 7,600 mg/L and sulfate levels generally range from 1000 to 5,000 mg/L (Hecla, 1998a), which are higher than the EPA Secondary Drinking Water Standards of 500 mg/L total dissolved solids and 250 mg/L sulfate.

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[2] Indiana Department of Environmental Management Office of Water, Tier I Human Health Noncancer Criteria; posted on an EPA webpage; https://www.epa.gov/glicearinghouse/human-health-noncarcinogen-fact-sheet-vanadium-human-health-noncarcinogen-fish, accessed 8/10/2020.

Table 1. Groundwater quality criteria for elements and radionuclides simulated in this report. Range of maximum measured values derived from seven onsite wells. [1 pCi/L = 37 Bq/m^3].

Component	EPA Primary Drinking Water Standard *	10 CFR 40 Appendix A	Proposed Durita Concentration Limits, Hecla, 1992	Maximum Values MW-8 to MW-14 (Hecla, 1998a, App. E)
Total Uranium	30 μg/L	-	-	5 to 30 μg/L
Radium-226/228	5 pCi/L	5 pCi/L	5 pCi/L	2.5 to 10 pCi/L
Thorium-230	None	-	60 pCi/L	3.0 to 6.9 pCi/L
Lead	0.015 mg/L	0.05 mg/L	-	-
Vanadium #	50 μg/L	-	-	-

  * www.epa.gov/dwstandardsregulations
  # California notification level for drinking water; not an EPA drinking water standard

4   Transport Scenario Pathways

Two transport scenario pathways for the Durita site are shown in Figure 3. The two pathways reflect end members of possible pathways consistent with the conceptualization of flow and transport at the site. The RESRAD-OFFSITE modeling considers the entire contaminated zone in each scenario transport pathway separately. In actuality, a combination of the two pathways is possible, with the contaminants split between the two. However, a summation of results is not appropriate because of spatial differences in the pathways. Rather, the results if both pathways were simulated concurrently would lie somewhere between the individual results of either transport scenario pathway if the plumes possibly overlap.

For Transport Scenario Pathway 1, staff assumed that there are no through-going vertical fractures in the Mancos Shale so that all the contamination would migrate laterally along the bedrock interface in the unconsolidated sediments in the model. The bedrock is assumed to have a low effective permeability, such that infiltrating water flows northward along the prominent paleochannel at the bedrock-unconsolidated sediment interface to the site boundary. This pathway was considered in the placement of earlier wells (Four Corners ER, 1977) along arroyos. The groundwater is then assumed to instantaneously flow vertically down a hypothetical fracture at the site boundary at which point the contamination reaches the designated uppermost aquifer. A hypothetical well is placed in RESRAD-OFFSITE immediately downstream of the hypothetical fracture at the site boundary. The important properties of this pathway are the hydraulic conductivity values associated with the unconsolidated sediments and distribution coefficient values that reflect the likely oxidizing conditions of the groundwater. The distance from the leach tanks to the designated uppermost aquifer below the site boundary along the bedrock-unconsolidated sediment interface is 494 m (1620 ft).

For Transport Scenario Pathway 2, staff assumed that vertical through-going fractures in the Mancos Shale are present and that they would instantaneously convey contamination vertically through the unweathered Mancos Shale to the designated uppermost saturated layer. Then, horizontal flow approximately northward towards the fence line is simulated based on the

hydraulic properties of the sandstone and other conductive layers in the lower Mancos Shale. Therefore, the unsaturated zone in the unweathered Mancos Shale is neglected based on the simplifying assumptions that flow and transport vertically through fractures in the unsaturated zone is fast and sorption of radionuclides on along fractures is negligible. The NRC staff notes that the potential to seep through faults/fractures to the aquifer below can occur anywhere along path from the source areas to the compliance location at the fence, but Pathway 2 conservatively assumes the through-going fractures occur beneath the leach tanks. The important properties for this pathway are the hydraulic conductivity values that reflect the conductive layers in the Mancos Shale and the distribution coefficient values that reflect the likely reducing condition of the groundwater system. A hypothetical well is placed in RESRAD-OFFSITE in the aquifer at the fence line, which is 494 m (1,620 ft) from the leach tanks.
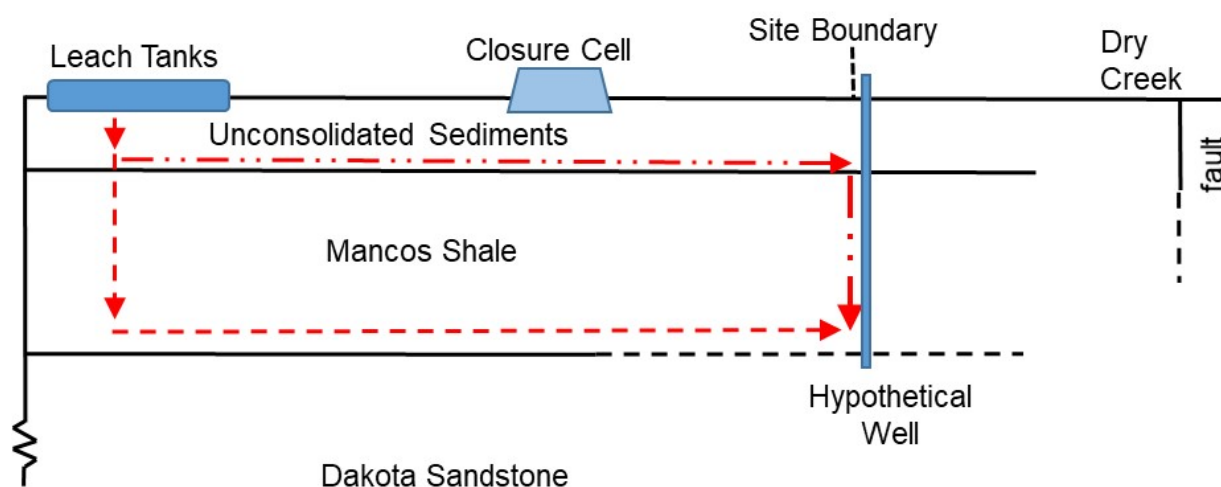

Figure 3.   Schematic of Durita Site conceptual model with Transport Scenario Pathway 1 (red dash-dot line) and Transport Scenario Pathway 2 (red dash line)

5   Source Term and Geochemical Behavior

Modeling of a potential release and migration offsite requires information on a source area and geochemical conditions. The source term includes the volume and concentration of radionuclides and any other contaminants that are important to consider in the modeling. Geochemical conditions primarily influence the sorption characteristics of each contaminant, and thus the release rate and the transport rate along pathways to offsite locations. Distribution coefficients reflect the distribution of a contaminant between the water and solid phases, and may also be called a partition coefficient or sorption coefficient.

5.1   Contaminant Selection

Radionuclides commonly found in relatively high concentrations in tailings from acid leach mills are radium-226, lead-210, polonium-210, thorium-230, and uranium (CDPHE, 2016; Robinson,

2004). Metals including barium, beryllium, cadmium, chromium, nickel, antimony, lead, mercury, silver, molybdenum and vanadium may be found in elevated concentrations, as may the nonmetals nitrate, cyanide, selenium and arsenic. Uranium and vanadium may also be in significant quantities in waste tailings even though they were the extracted metals in the heap leach processing because efficiency of extraction is not 100 percent.

The starting point for identifying the constituents to use in a source term for the scenario modeling in this report is the list of the constituents of concern listed in the CRR. CDPHE (2016) described the process used to determine the constituents of concern to include in the license for the detection monitoring program. The amount of radium-226 remaining in the Durita waste rock inside the leach tanks is recorded on the site monument. Information on other contaminants in tailings of the leach tanks is less directly accessible. CDPHE (2016) stated that seven agglomerator (feed tailings) samples from Durita were analyzed for the non-radiological elements arsenic, cadmium, lead, molybdenum and selenium and the radiological elements thorium-230 and radium-226. However, analyses of the feed tails did not include uranium or vanadium (CDPHE, 2016, 1998). Radium-226, thorium-230, arsenic, lead, and selenium were detected in the feed tails, while cadmium and molybdenum were not. CDPHE (2005c) provided data on molybdenum of 50 mg/kg (ppm) tailings and 30 mg/kg (ppm) background, selenium of 4.9 mg/kg (ppm) tailings and 0.52 mg/kg (ppm) background, and arsenic of 26.7 mg/kg (ppm) tailings and 9.82 mg/kg (ppm) background. AK Geoconsult (1993) provided results of analyses of samples from the evaporation ponds that held the raffinate (effluent with uranium and vanadium removed). The results for arsenic ranged from 23 to 74 mg/kg (ppm); results for selenium and chromium were not provided in AK Geoconsult (1993) and the results for radium-226 and thorium-230 are discussed in the next subsection. In addition, analyses of the Naturita tailings should provide an adequate representation the Durita waste tailings because the tailings piles from Naturita were moved to Durita for processing in 1978 and 1979. Values of concentration for the Naturita waste tailings indicate that arsenic and lead range from 40 to 60 mg/kg (ppm) and selenium at about 0.5 mg/kg (ppm) (DOE, 1981). At these levels, arsenic, lead, and selenium concentrations are not sufficiently high to warrant inclusion in the source term for modeling because these levels would not lead to significant offsite contamination compared to background levels and water quality standards. The licensee retained arsenic and selenium as indicator elements for a groundwater monitoring program because they would be the first contaminants to appear at monitoring wells based on sorption characteristics.

An additional consideration is that uranium and vanadium contamination were found in the alluvial aquifer below the waste impoundment area at Naturita (Davis and Curtis, 2003), thus suggesting that vanadium could also be included in the Durita source term for modeling. Contamination of the alluvial sediments below the tailings piles (now removed) at Naturita is not, however, analogous to Durita in terms of potential for release because the former site was unlined and the latter site utilized a compacted clay liner.

Based on the discussion above, the primary contaminants that are considered for incorporation into the source term for Durita modeling are uranium-238, uranium-234, thorium-230, radium-226, and vanadium. The following subsections discuss estimates of concentration and

estimates of the source term volume. The last subsection provides a discussion of the geochemical behavior of the contaminants selected for modeling in the modeled Durita groundwater environment.

5.2   Contaminants Remaining in Impoundments

The monument emplaced at Durita indicates that 274 curies (10.1 TBq) of radium-226 are contained in 700,000 tons (635,000 Mg) of waste rock in the leach piles and closure cell (Figure 4). The NRC Office of Nuclear Materials Safety and Safeguards Procedure SA-900, Termination of Uranium Mill Licenses in Agreement States, recommends that a CRR (e.g., CDPHE, 2016) contain a description of the radioactivity in waste impoundment piles that includes curies of radium-226, thorium-230, and total uranium. CDPHE (2016), while citing Procedure SA-900, did not mention the number of curies nor a description and basis for the radioactivity from the leaching operation that remains on the site.

The NRC staff found an unnumbered table with radium-226 analyses of tailings samples in Appendix D of Volume II of the Final Reclamation Report (Hecla, 1991, Volume 2, 152nd page). The average concentration of the 10 relevant samples (leach tank waste rock) multiplied by the total number of tons of waste rock in the three leach pits turns out to be 274 curies (10.1 TBq) of radium-226, which staff infers is the supporting basis for the value inscribed on the monument.
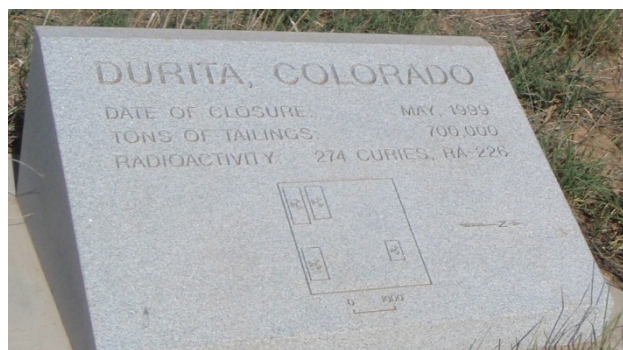

Figure 4. Monument located at Durita Site

The NRC staff derived confirmatory estimates of radium-226 plus estimates of total uranium and thorium-230 at Durita based on the likely processing efficiency at Durita, and on literature on the Naturita mill site, which was the source of the uranium tailings that were processed at Durita. The leach tanks and closure cell are treated separately in this report.

Leach Tanks

Most of the radioactivity in waste rock at mill tailings and leaching sites is due to the daughter products of the uranium from the original ore. Naturally occurring uranium ore contains mostly uranium from the uranium-238 decay chain (99.3%)[3]; uranium-238 is the head of the uranium decay chain that includes uranium-234. The other natural isotope, uranium-235 (0.7% in natural rocks), is the head of the actinide decay chain. In general, removal of uranium in processing leaves approximately 85% of the original radioactivity (DOE, 1977, page 1-7; Robinson, 2004, page 9). A large percentage of the radioactivity comes from the daughter products that were

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[3] http://www.nrc.gov/reading-rm/basic-ref/glossary/uranium.html

under secular equilibrium[4] with uranium-238 in the ore. Ore deposits in the Uravan Mineral Belt were generally found to be in secular equilibrium (Davis and Curtis, 2003, page 20). Secular equilibrium is a reasonable assumption for the time frames of many ore deposits, but not for the time frames of milling operations and extractions. Addressing only the uranium-238 series because of its predominance in nature, the amount of each uranium-238 daughter in the decay chain can be estimated by assuming secular equilibrium in the original ore and ignoring the small changes after disequilibrium was created by uranium extraction, especially given the time that has transpired since milling activities at the site compared to the half-lives of the radionuclides.

The amount of uranium remaining in the waste tailings can be estimated using extraction efficiencies at both Naturita and Durita. After processing the ore, most of the daughter products remain in the waste rock along with a portion (1 minus the efficiency of removal) of the uranium and vanadium. The significant exception for daughter products remaining in the waste tailings is thorium because it behaves similarly to uranium in the acid leaching phase of processing. At both Naturita and Durita, some portion of the thorium is leached during the acid stage of processing and ends up in the evaporation ponds. Thorium is not expected to migrate in the natural groundwater system nor to be released to the groundwater system from the closure cell because it readily precipitates in solutions above approximately a pH of 3 (see, e.g., EPA, 1999a). The daughter products of thorium, however, may become important during the 1000-yr performance period.

Estimates of the quantity in curies of selected radionuclides are provided in Table 2. The estimates are calculated based on the following constraints:
- 704,000 tons (639,000 Mg) of 0.3% uranium oxide concentrate (U_3O_8) ore processed at Naturita; 700,000 tons (635,000 Mg) shipped to Durita
- For the amount of residual uranium-238 in waste tailings after processing at Durita:
  - Uranium-238 in ore is approximately 84% of naturally occurring uranium oxide concentrate ore by weight.
  - Efficiency in extracting uranium at the Naturita mills was 77 to 79 percent (Albrethsen and McGinley, 1982).
  - Generic efficiency for heap leaching operations similar to that at Durita is 60 to 70 percent. A lower extraction efficiency for uranium of 44 percent, however, is calculated for the Durita operations using information from the 1979 annual report (DOE, 1981) to derive pounds of uranium oxide concentrate recovered at Naturita and Durita, processing costs per ton of tailings, and recovery costs per pound of uranium oxide concentrate.

The amount of daughter products in tailings at Durita is based on secular equilibrium of uranium-238 in the original ore with the amount of uranium-234 adjusted to account for uranium extraction. The estimates of curies in the leach tanks in Table 2 (column 3), however, should be considered a maximum amount. Most of the radium-226 can reasonably be assumed to remain

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[4] In nuclear physics, secular equilibrium is a situation in which the quantity of a radioactive isotope remains constant because its production rate (e.g., due to decay of a parent isotope) is equal to its decay rate.

in the tailings because radium is not extracted by the acid treatment that separates uranium from the milled ore. Some portion of the thorium-230 should be carried through the effluent and delivered to the evaporation ponds. Because of geochemical similarities, acid extraction of uranium would also extract thorium from the tailings. Much of the uranium would then be separated from the uranium-rich solution in the next processing step, but the thorium would be discharged in the low pH solution to the evaporation ponds. The presence of thorium-230, and to a lesser extent uranium, in grab samples from the evaporation ponds at Durita (AK Geoconsult, 1993) supports this conceptualization. Therefore, estimates of uranium and thorium-230 in the leach tanks in the Table 2 are conservative. The large concentrations of thorium-230 and lesser concentrations of uranium in the Durita evaporation pond samples are similar to what is seen at other heap leach sites (Landa, 2004) where the raffinate solutions are disposed separately from tailings. Thorium-230 and uranium concentrations in the solidified pond material are discussed further in the next section, which focuses on the closure cell.

The estimate in Table 2 for radium-226 is larger by a factor of approximately two compared to that recorded on the Durita monument. To check if the Table 2 estimate is reasonable for Durita, data from literature on the Naturita tailings were utilized. Dames and Moore (1978, page 7) provided an estimate of the radium-226 concentration in the Naturita tailings between 728 and 873 pCi/g (27 and 32 Bq/g). If radium remains in the tailings after processing, rather than being extracted during heap leaching, the radium-226 concentrations in the Naturita and Durita tailings should be approximately the same as the original ore. This assumption is supported by measurements of raffinate and evaporation pond samples at Durita that indicate radium-226 concentration is less than 1 percent of that in the waste tailings. The assumption

Table 2. Estimates of activities and concentrations of radionuclides in the waste rock in leach tanks at Durita. Leach tank estimates for curies and concentration neglect the transfer of radionuclides to the evaporation pond and closure cell. [1 curie = 0.037 TBq, 1 pCi/L = 37 Bq/m^3, 1 mg/kg = 1 ppm].

Radionuclide	Activity, Curies		Concentration	Concentration	
	Value Listed on Monument	Estimate for this Report for Leach Tank Plus Cell	Estimate Assuming All Activity Is in Leach Tank	Evaporation Pond Only, Measured [#]	Solidified Pond Material Only, Estimate for Cell
Uranium-238	-	71.0	112 pCi/g	12 pCi/g	8.0
Uranium-234	-	70.4	111 pCi/g	17 pCi/g	11.3
Uranium-235	-	0.51	0.8 pCi/g	0.8 pCi/g	-
Thorium-230	-	552	870 pCi/g	1126 pCi/g	751 pCi/g
Radium-226	274	549	864 pCi/g	7.4 pCi/g	4.9 pCi/g
Total Uranium	-	142	329 mg/kg	56 mg/kg	-
Vanadium	-	-	1,980 mg/kg	-	-

[#] Data from grab samples reported in AKGeoconsult (1993) Table 2; U isotope data for a different set of samples reported in Hecla (1991) Table 4.

that radium-226 remained in the leach tank at Durita is also supported by analyses at other, similar operations where the raffinate solution is disposed of separately from the crushed ore (e.g., Landa, 2004). Based on the Naturita tailings concentration, and accounting for the 700,000 tons (635,000 Mg) of tailings that were transported from Naturita to Durita, there should have been a range of 460 to 565 curies of total radium-226 in the Durita tailings. This range corresponds well with the estimate in Table 2 [549 curies (20.3 TBq)], and is a factor of two higher than the value recorded on the Durita monument.

Comparisons with Other Title II Sites

Table 3 provides context for the estimate of curies and concentration for radium-226 at Durita in comparison to Title II sites released to DOE. The estimate of radium-226 concentration at Durita is at the high end compared to other sites. The number of curies of radium-226 at Durita is comparable to other sites, except for Bluewater.

Table 3. Comparison of radium-226 in Durita tailings compared to Title II sites, based on information available at https://energy.gov/lm/sites/lm-sites.
[1 curie = 0.037 TBq, 1 Ton = 0.907 Mg, 1 pCi/g = 0.037 Bq/g, 1 ft = 30 cm]

Site	Activity Radium-226, Curies	Mass of Tailings, Tons	Concentration Radium-226, pCi/g	Base of Tailings Pile
Durita CO	274 (549) [&]	700,000	430 (864) [&]	1-ft thick clay liner, over thin shale layer
Bluewater NM	11,200	23,000,000	537	Mostly basalt
Edgemont SD	526	4,000,000	145	Thick shale
Shirley Basin WY	974	6,300,000	170	Sand and shale layers
Maybell West CO	96	1,975,000	54	1-ft thick clay liner
Sherwood WA	476	3,075,000	171	Synthetic liner
L-Bar NM	505	2,100,000	265	Thick shale

[&] First value is that reported on the monument; value in parentheses is estimate derived as a conservative input for modeling in this report

Closure Cell

The closure cell was constructed in a portion of the evaporation ponds area on the north side of Mancos Hill. A 1-ft (30-cm) thick compacted clay liner was constructed from clayey material at the site and embedded in the Mancos Shale. The closure cell is comprised of contaminated material from the evaporation ponds, raffinate ponds, and incidental contaminated material from across the site. The ultra-low pH (e.g., < 3) of the pond material was stabilized by adding carbonate-rich Mancos Shale at a 1:0.5 pond material to rock ratio by volume. The closure cell was covered with a 1-ft (30-cm) thick compacted clay layer and a 5- to 8-ft (150- to 240-cm) thickness of additional sediment cover to act as a radon barrier.

The volume and types of contaminants of the material in the closure cell is not quantitatively known. Ten samples of the material from the raffinate and evaporation were analyzed in

preparation for mixing with carbonate-rich shale to neutralize the acidity prior to closure. The averages reported for the raffinate and evaporation pond material are approximate because they are based on a small number of non-representative grab samples, some of which include only one of the three phases in the ponds (solution, gelatinous or salt crystal; AK Geoconsult,1993; Hecla, 1991).   Average measured concentrations of selected radionuclides are shown in Table 2 for qualitative comparisons only. Because the volumes are not known for either the different phases of pond material or the other contaminated material placed in the closure cell, the amount of different radionuclides in the closure cell cannot be reliably estimated. The solution and gelatinous materials (i) were extremely acidic with pHs ranging from 1.8 to 3.1 (AK Geoconsult, 1993), (ii) had high sulfate due the acid leaching process, and (iii) had high chloride due to the solvent extraction process. Thorium readily stays in solution at these low pH conditions. Thorium is similar to uranium geochemically and, as such, is extracted from the tailings when in contact with the low pH sulfuric acid, but is not extracted from the effluent in the second step of processing.

An excerpt of an engineering report, sent to NRC as a facsimile (CDPHE, 2005b), contains information that may support the magnitude of thorium-230 concentration contained in the evaporation ponds. Solidified pond material contains two parts evaporation and raffinate pond material mixed with one-part carbonate-bearing Mancos Shale, by volume. Accounting for the mixture, the thorium-230 and radium-226 concentrations in Table 2 would be adjusted to 751 pCi/g and 4.9 pCi/g (27.8 Bq/g and 0.18 Bq/g) for the solidified pond material emplaced in the closure cell. CDPHE (2005b; incomplete document) contains an estimate of the solidified pond material of 743 pCi/g (27.5 Bq/g) for thorium-230 and 6.4 pCi/g (0.24 Bq/g) for radium-226, although the derivation or analyses supporting these values was not provided.

The close comparison is either fortuitous or indicates that the evaporation pond grab samples are representative, and that some significant fraction of the thorium-230 was shifted to the closure cell from the leach tanks. Whereas a value for the concentration of thorium-230 can be estimated for the solidified pond material, the volume of solidified pond material in the closure cell is not similarly constrained. Furthermore, an unknown volume of other contaminated material from the site was emplaced in the closure cell. Therefore, the total amount of curies of thorium-230, radium-226, and uranium isotopes in the closure cell are not known. The distinction between radioactivity in the closure cell and the leach tanks, however, is not important for the modeling described in Section 6 because the source term is conservatively simplified to one contiguous volume. This is conservative because (i) the barrier components of the closure cell would be expected to perform better than those of the leach tanks, (ii) the contaminated material put into the cell was chemically neutralized, and (iii) separately modeling the leach tanks and cell spreads out the contamination in space and time, which would lead to lower peak concentrations at the boundary than the modeled approach.

Non-Radiological Contaminants

CDPHE (2016, 1998) considered arsenic, lead, cadmium, molybdenum, and selenium as possibly present in the feed tailings. As discussed in Section 5.1 Contaminant Selection, none

of these metals were found in significant concentrations that would warrant inclusion in the modeling. Vanadium, however, is considered because it was found in the contaminated groundwater at Naturita, and because it was one of the metals extracted for purification and sale at Durita. Vanadium concentrations in the waste tailings were not provided in available documents, but estimates are derived here based on (i) ore concentrations, (ii) Naturita extraction efficiency, and (iii) an estimate of the Durita extraction efficiency. Like uranium, vanadium may be present in both the leach tank and closure cell.

The vanadium concentration in the ore was 1.8 percent as vanadium pentoxide (V_2O_5) (DOE, 1981, 1977), of which elemental vanadium is 56 percent by mass fraction. The mass of uranium-vanadium ore at Naturita was approximately 704,000 tons (639,000 Mg) (DOE, 1982). The extraction efficiency for vanadium at Naturita was 68 percent (DOE, 1982, pp. A-24 to A-28). The extraction efficiency at Durita is not known, but is estimated by assuming the same efficiency ratio between Naturita and Durita for uranium. Combining this information, the vanadium concentration in the waste at Durita is estimated to be 1,980 mg/kg (ppm). DOE (1981, p.3-17) reports an analysis of auger samples of the waste tailings at Naturita as high as 3000 mg/kg (ppm); however, this is maximum value and it does not include processing and extraction of vanadium at Durita.

Vanadium more readily sorbs onto to solids and does not migrate as quickly in groundwater compared to uranium. The distribution coefficient for vanadium is discussed in the subsection "Behavior of Radionuclides in Groundwater System" of Section 5, below.

Assumptions for Source Term

Estimates of concentration and source volume are needed for the RESRAD-OFFSITE modeling described in Section 6. Given the uncertainties described in the subsections above, a simplified approach is taken for the source term. All of the radionuclides in the leach tanks and closure cell are assumed to be contained in a single leach tank source volume. Given neutralization of the solidified pond material with carbonate-bearing Mancos Shale, and the ready precipitation of thorium-230 in environments with a pH>3, thorium-230 is not likely to migrate significantly from the closure cell. Thorium-230 is retained in the analysis because one of its daughters, radium-226, is more mobile in the environments of the groundwater transport pathways at Durita. The single volume of the source term used in RESRAD-OFFSITE retains the average thickness of the waste tailings in the leach tanks, and retains the footprint that reflects the combined area of the three leach tanks. Conservatively, the distance from the hypothetical source term volume to the compliance location uses the minimum distance between the closest leach tank (LT-202) and the fence line at the property boundary. The concentrations used in the RESRAD-OFFSITE modeling are from the Table 2 column with the leach tank estimate of concentration (4th column). The combination of source volume and concentration provides the total activity at the site.

5.3　Behavior of Radionuclides in Groundwater System

The mobility of contaminants is often modeled using distribution coefficients that describe the amount of the contaminants that stay in the water phase compared to the portion that remains attached to the solid phase (e.g., adhering to mineral grain surfaces). For the groundwater transport portion of performance assessments, selection of lower values for distribution coefficients is conservative to site performance because more of the radionuclides would remain in the groundwater phase and would therefore migrate offsite faster and at higher peak concentrations. The term distribution coefficient is used in this report instead of sorption coefficient.

The distribution coefficient reflects sorption processes (ion exchange, electrostatic forces, and surface complexation), mineral precipitation reactions, and the formation of solid solutions; the latter includes both direct co-precipitation and incorporation into previously formed solids. The commonly used laboratory batch test and field-based measurement used to estimate distribution coefficients (K_d) for a site do not delineate which of these processes dominate the measurement. This means that an aggregation of processes is included in each K_d value. The distribution coefficient values apply to the element and do not change for each isotope of that element, because the chemical behavior for each isotope is essentially the same.

Site-specific information is generally recommended for estimating the distribution coefficients because of effect of environmental conditions on the various processes that affect contaminant migration. Mineralogy, pH, alkalinity, Eh, groundwater constituents, and interactions with other solutes all could affect the processes controlling distribution between the solid and liquid phases for each radionuclide. In addition, K_d values vary spatially and temporarily, thus making simulations with a single K_d-based model problematic (Zhu, 2003). Because of this complexity and condition-specificity, many references state that distribution coefficient values measured for site-specific conditions are absolutely essential (e.g., EPA 2004, page 5.18). As a result, with limited site-specific information available for the Durita site, distribution coefficients are prime parameters for uncertainty analysis or for conservatism.

The approach taken here was to use literature surveys of distribution coefficients to select conservative values for Durita. For the radionuclides and vanadium, these surveys included Sheppard and Thibault (1990), McKinley and Scholtis (1993), EPA (1999a, 2004, 2005), Mitchell et al. (2013), IAEA (2010, 2014). There is overlap of the data sets used in each of these references. The surveys generally group results according to ancillary parameters of soil texture, organic content, and pH to reduce uncertainty in selecting a value for a site. The relevant environmental conditions and textures at Durita for estimating distribution coefficients are neutral pH, oxic to reducing redox conditions[5] depending on the transport pathway, and soil textures of clay to silt loam soils for the unconsolidated sediments, and clayey to silty sands for the water conductive units of the Mancos. The organic content of the unconsolidated soils at Durita is likely low because of the small amount of vegetation. Table 4 provides a context for

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[5] Environmental conditions that refer to the presence or depletion of dissolved oxygen.

the values selected as inputs for the Durita site. Higher values of distribution coefficients correspond with larger retardation factors, which leads to slower transport of the contaminant.

Table 4. Distribution coefficient values in L/kg reported as geometric mean value. Uranium(VI) refers to uranium in a valence state of +6, which dominates in oxic environments, and uranium(IV) refers to the +4 valence state for reducing conditions.

Element	RESRAD Default	Yu, et al. (2015)			IAEA (2010) for soil	Selected for Durita
		Sand	Loam	Clay		
Radium	70	3,100	1,100	13,000	2500	70
Thorium	60,000	700	18,000	4,500	3300	1700 *
Uranium(VI)	50	110	310	28	740	15
Uranium(IV)	-	-	-	-	-	100
Vanadium	935	180	-	-	300	50

* For pH range 5 to 9, all soil textures (EPA, 1999b)

The bases for the values of the distribution coefficients selected for Durita in the last column of Table 4 are described in the following subsections.

Distribution Coefficient for Uranium

Uranium has two relevant oxidation states (IV and VI) that predominate depending on the redox conditions. Illustrating the effect of redox conditions on the distribution coefficient, (i) uranium readily precipitates in roll-front uranium deposits at redox fronts that transition from oxic to reducing conditions, and (ii) the use of permeable redox barrier remediation technology. Black shales like the Mancos Shale contain reduced iron (e.g., pyrite) and carbon (DOE, 2011) indicating that reducing conditions should be prevalent in the unweathered bedrock at Durita. Uranium(IV) is the dominant species in reducing and sub-oxic (sulfate may be present) environments and tends to form sparingly soluble precipitates that commonly control uranium concentrations in groundwater (EPA, 1999a; Davis and Curtis, 2003). Distribution coefficients for uranium(IV), therefore, are not readily available in the literature. Whereas there may be little uncertainty that the distribution coefficient for uranium(IV) in reducing environments is large, there may be uncertainty in the conversion of uranium(VI) in the tailings to uranium(IV) in the anoxic environment of the unweathered Mancos Shale. The reduction of uranium(VI) to uranium(IV) is thermodynamically favored, but it may be kinetically hindered (Zhao et al., 2016), though the time scale for reduction may be significantly shorter than the 1,000-yr performance period. Considering the kinetic uncertainty, a K_d value of 100 L/kg is used as the input for the unweathered Mancos Shale in Transport Scenario Pathway 2 based on the screening value for a pH range of 5 to 9 in the MEPAS code (Multimedia Environmental Pollutant Assessment System), as described in EPA (1999b, Table 3.2). This uncertainty covers the possibility of uranium(VI) remaining in the groundwater, with incomplete reduction to uranium(IV).

Uranium mobility in the unconsolidated sediments and weathered Mancos Shale should be based on uranium(VI) and its complexes as the dominant species because oxidizing conditions are expected. The K_d for uranium under oxidizing conditions, however, is difficult to estimate

without site-specific analyses or measurements. Compilations of distribution coefficient studies for uranium use pH or soil texture as ancillary parameters to reduce uncertainty. However, uncertainty of uranium K_d values within the soil texture or pH categories still spans a large range; i.e., up to four orders of magnitude. For example, the uranium K_d range in EPA (1999a) for 7<pH<8 is 63 to 630,000 L/kg, and in Sheppard and Thibault (1990) for loam is 0.2 to 4500 L/kg and for clay is 46 to 395,100 L/kg. Rather than using the minimum measured or estimated for any site in the compilations, this modeling evaluated uses a refined delineation based on geochemical and textural comparisons with other sites in the arid western United States. Conclusions from uranium(VI) sorption studies indicate that the increased mobility of uranium(VI) in aquifers with neutral to alkaline pH values is caused by the formation of the uranyl-carbonate complexes (Fox et al., 2006; Serne, 2007; EPA, 1999a), including the study at the Naturita site (Davis and Curtis, 2003). Therefore, in addition to soil texture and pH mentioned above, uranium(VI) mobility depends on alkalinity. The features of Durita to consider in comparison with other sites are: (i) a high clay and silt content in the unconsolidated sediments, (ii) a sulfate dominated groundwater with low to moderate alkalinity, and (iii) pH of the groundwater in the range 7<pH<8.

Studies at several semi-arid sites in the western U.S. illustrate the connection of uranium mobility, uranyl-carbonate complexes, and low values for distribution coefficients. At Hanford, Gamerdinger et al. (2001a) found K_d values of 0.77 and 1.54 L/kg in for the sands (99 percent sand) and silty sands (80 percent sand) that were the dominant sediment textures at the site. Lower values were found at Hanford when high flow rates were compared with low flow rates in laboratory experiments, which Gamerdinger et al. (2001a) attributed to a kinetic rate effect on the sorption process. Gamerdinger et al. (2001b) found higher K_d values averaging 4.77 L/kg for silt loam (35 percent sand, 56 percent silt) at Hanford. The pH of 8.4 for the groundwater at the Hanford site indicates that carbonate complexes would dominate in the groundwater. Um et al. (2010) further illustrated the difference between coarse and fine sediments below the tank farm at Hanford. They reported uranium K_d values of less than 2 L/kg for coarse sediments and from 832 to 983 L/kg for fine-textured sediments at Hanford. For the alluvium and interbed sediments at Idaho National Laboratories, Prikryl and Pickett (2007) suggested the best estimate for the uranium K_d is 1.6 L/kg. The dominant anion in the groundwater at the Idaho National Laboratory site is bicarbonate. At Naturita, which is the originating site of the waste tailings at Durita, the calculation of uranium K_d values based on the groundwater chemistry in the alluvial sediments ranged from 0.29 to 22 L/kg (Davis and Curtis, 2003). There was a bimodal distribution of uranium K_d values at the Naturita site, with low K_d values associated with portions of the uranium(VI) groundwater plume containing high alkalinity and high concentrations of dissolved uranium(VI). Higher K_d values at Naturita were associated with low concentrations of dissolved uranium(VI) and low alkalinity. The alkalinity at the higher portion of the bimodal range was approximately 550 to 1,220 mg/L as bicarbonate at Naturita. The Naturita site alluvial sediments are dominantly sands, gravels, and silty sands. In addition, the uranium and vanadium groundwater contamination at Naturita was found under the western side of the former tailings pile where the carbonate-leached tailings were placed. Little contamination and low alkalinity were found on the eastern side of the Naturita site, closer to the river, where only acid-leached tailings were placed. The carbonate-leached tailings were

reprocessed later at another site, not at Durita (Davis and Curtis, 2003). The common features at these western U.S. sites with low uranium K_ds are carbonate or bicarbonate groundwater and coarse sediments (sands, silty sands, and gravels).

Features and groundwater chemistry at Durita are more consistent with that of the other sites in the Western U.S. where moderate to high values of Kd were reported. Durita is a gentle hillslope environment where eolian and colluvium sediments dominate. Based on borehole logs, silty and sandy clays dominate the unconsolidated sediments at the Durita site (CDPHE, 2015). The groundwater chemistry for unconsolidated sediments in the Coke Oven Basin average 512 mg/L as carbonate (624 mg/L as bicarbonate) and a pH of 7.5 (Fox, 1982). In the lower Mancos Shale at the site, which also may have some relevance to the weathered Mancos Shale, the average alkalinity for wells proximal to Transport Scenario Pathway 1 is 510 mg/L as bicarbonate and the pH is 7.5 (Hecla, 1998a). No measurements of alkalinity or pH were made in the unconsolidated sediments directly on the Durita site because no saturated zones were identified. Whereas the characteristics of the tailings are similar at the two sites (i.e., the tailings were moved from Naturita to Durita), the difference in sedimentary environments between the sites warrants a K_d value for Durita larger than that estimated at Naturita. The alluvial sand and gravel sediments at Naturita likely sorb less readily than the clay- and silt-dominated eolian and colluvial sediments at Durita. A K_d value of 15 L/kg is conservatively selected for the unconsolidated sediment because: (i) it is half the value of the minimum value (the clay value) shown in Table 4 based on the data inputs document supporting RESRAD (Yu, et al., 2015), and (ii) Durita should have a larger uranium K_d than found at the western U.S. sites described above because of much higher clay and silt content, low to moderate alkalinity, and near neutral pH. Sensitivity analyses in Section 7 assess the effect of smaller values (i.e., 10 and 5 L/kg) for the uranium K_d.

Distribution Coefficients for Other Elements

Distribution coefficients for three other elements are considered here: radium, thorium, and vanadium. For radium, the same value for the distribution coefficient is used in both transport scenario pathways for unconsolidated sediments and for Mancos Shale. Radium exists only in the 2+ oxidation state, thus indicating that the redox state of the groundwater is not directly a factor in the value of the distribution coefficient. Other ancillary parameters used to reduce uncertainty for the radium K_d, such as soil texture, pH, or other geochemical factors, do not result in differences between categories (Vandenhove, et al., 2009), though the small sample size of available measurements may be a factor in the significance analysis. The EPA (2004) compilation for radium shows a range from 1,262 to 530,000 L/kg. IAEA (2014) and EPA (2004) state that precipitation of radium-bearing minerals is readily possible for sulfate- or carbonate laden groundwater with radium entering the solid phase preferentially compared to the other 2+ cations, following the order radium>barium>strontium>calcium>magnesium (EPA, 2004; Serne, 2007; IAEA, 2014). At the Durita leach tanks, high sulfate values due to sulfuric acid treatment may prevail, and possibly lead to lower release rates. Distal to the Durita tanks, along the transport pathways, the high background sulfate values of the groundwater based on the well monitoring data may enhance precipitation of radium-bearing sulfate minerals. The mean

sulfate concentration in the seven groundwater wells at Durita is 1938 mg/L with a maximum value of 5,000 mg/L based on data in Hecla (1998a). As noted above, high values of sulfate would lead to higher precipitation rates of radium-bearing sulfate minerals compared to lower values of sulfate in the groundwater. Higher mineral precipitation rates lead to larger K_d values. The default value of 70 L/kg in RESRAD for the radium K_d was conservatively selected for Durita because it is smaller than values reported from other sources recorded in Table 4.

Thorium is redox-stable and its distribution coefficient value is likely high (Mitchell et al 2013; Amayri, et al. 2016). The large K_d for thorium is not because it sorbs readily, but rather, because its concentration in solution is generally controlled by the low solubility of its primary minerals (EPA, 1999a). Under typical groundwater conditions, thorium readily precipitates at concentrations $>10^{-9}$ M (EPA, 1999a). For concentrations $<10^{-9}$ M thorium, the EPA (1999a) compilation for thorium shows a range from 1,700 to 170,000 L/kg. The RESRAD-OFFSITE default K_d value of 60,000 L/kg reflects thorium's lack of mobility in groundwater systems. In a claystone similar to the Mancos Shale, Amayri et al. (2016) estimated a K_d value of 29,000 L/kg based on laboratory measurements under a variety of conditions. Thorium is mobile when pH <3. For a pH 5 to 8 range, thorium in solution would be controlled by low solubility of thorium minerals rather than by sorption onto solids (EPA, 1999a, Mitchell, et al., 2013). The groundwater at Durita falls in the pH range of 7 to 8. The smallest thorium K_d value in Table 4 is for a sand, which is not appropriate for the loam sediments at Durita, but was added to the table to provide context. For simulations in Section 7, the K_d value of 1,700 L/kg is conservatively selected for Durita based on the low end of the range provided in EPA (1999a).

For metals, only vanadium was expected to potentially have high concentrations in the waste tailings or closure cell. Each of the cited compilations above only had a small sample set for vanadium from which to estimate a mean and range for distribution coefficient. At Naturita, the spatial extent of vanadium in the contaminated groundwater is much smaller than that of uranium, which indicates that the distribution for vanadium is larger than that of uranium (Davis and Curtis, 2003). To be conservative, vanadium as a contaminant is retained and assessed in the modeling for Durita. A conservative (low) value of 50 L/kg is selected for vanadium as listed in Table 4.

5.4   Simplification of Source Term for Modeling

The source term for the modeling in the following sections is simplified to a single volume represented by the footprint and tailings thickness of the leach tanks. Whereas the leach tanks are further from the compliance location than the closure cell, most of the uranium and radium are in the leach tanks. Uranium and radium-226 are the radionuclides expected to reach the compliance location first. Thorium-230 may be more prevalent in the closure cell, but it is not mobile in natural environments (see Section 5.3). The use of a single source for the site avoids the uncertainty in dividing the total radionuclide inventory between the leach tanks and closure cell; the source term inventory in Table 4 is an estimate for the entire site. All the uranium, radium, and thorium in the leach tanks and the closure cell are conservatively lumped into the single source term at the leach tank location in the modeling described in Section 6.

To support the use of a single source term at the leach tank location, a simulation is performed for the closure cell alone to confirm that the source inventory for the closure cell offsets the shorter distance to the compliance location; i.e., that the contamination reaching the compliance is substantially smaller than that from the leach tanks. Whereas there is uncertainty in the contaminants in the closure cell, the quantity and concentration appears to be smaller than in the leach tanks based on analyzed samples from the raffinate and evaporation pond residue in combination with the mixing of clean waste rock with the residue to geochemically stabilize the pond material prior to emplacement in the closure cell as discussed in Section 5.2; thorium is likely the exception. There were no measurements reported for thorium in the leach tanks, but high levels were found in raffinate and evaporation pond material (AK Geoconsult, 1993). An unknown amount of thorium was leached from the tailings waste and transferred to the evaporation ponds during processing, and later emplaced in the closure cell as part of the chemically neutralized solidified pond material. Chemically, this makes sense because at ultra-low pH, such as that found in the sulfuric acid processing and as measured in the ponds, thorium is mobile. The material from the raffinate and evaporation ponds was neutralized by mixing the waste with carbonate-bearing Mancos Shale as part of the emplacement of material in the closure cell. With a higher pH, as associated with neutralization by the carbonate in portions of the Mancos Shale, thorium would precipitate and would no longer be mobile in the water phase. Estimates of uranium concentration in the evaporation pond are significantly lower than in the leach tanks; concentrations in the closure cell would be even smaller due to mixing of solidified pond material with uncontaminated shale. Estimates of radium concentration are very low in the closure cell compared to that in the leach tank. In addition, ingrowth of radium from the decay of thorium-230 would not peak for more than 9,000 years. Even if all the thorium in the tailings migrated during processing to the evaporation ponds and now resides in the closure cell, the concentration of radium-226 at 1,000 years due to ingrowth is calculated to be approximately 300 pCi/g; however, ingrowth of radium-226 is not instantaneous, but rather builds up over time and thus has little effect on contamination reaching the compliance location during the 1,000-yr performance period.

The source term for the closure cell is simulated using the footprint of the cell and estimated average thickness of 6.2 m (20.5 ft), the distance of 94 m (310 ft) from the edge of the cell to the compliance location, and the inventory listed in the last column of Table 2. For the simulation results, none of the radionuclides reach the compliance location in the first 1,000 years. The radium-226 peak value of 1 pCi/L occurs at approximately 54,000 years, which is later than the leach tanks peak because it builds up by ingrowth from the decay of Th-230 and its other daughters. The uranium concentration peaks at less at 5 percent of the peak value of the result for the leach tanks presented in Section 7. Therefore, modeling a single source at the leach tanks is conservative compared to separately simulating the leach tanks and closure cell because the latter would spread out the contamination and reduce the peak values reaching the compliance location.

6 Model and Transport Parameters

This section includes a brief description of the code and model used to calculate concentrations at the compliance location. Also included are descriptions of selected parameter values and their bases. Detailed discussion is based on their importance to performance in combination with uncertainty of their input values. Some important parameters, such as distribution coefficients, have been described in previous sections. Those discussions are not repeated here.

6.1 RESRAD-OFFSITE Model

RESRAD-OFFSITE version 3.2 is used to estimate concentrations of radionuclides in the groundwater at the compliance location for each of the transport scenario pathways. Whereas RESRAD-OFFSITE is a stochastic code for calculating dose, it is used in this report for deterministic calculations of concentration in the groundwater system at designated locations. The capability in RESRAD-OFFSITE to convert concentration to dose for different exposure scenarios is not used here. Nor is the capability used for surface erosion and overland or atmospheric transport. The relevant user guide for RESRAD-OFFSITE version 3.2 is NUREG/CR-7189 (NRC, 2015).

The primary components of RESRAD-OFFSITE needed are the (i) source release; (ii) segments of the groundwater transport pathway; and (iii) changes to the concentration along each segment that address radioactive decay and ingrowth of daughter products, sorption, and dispersion. The segments of the groundwater transport pathway include water flux through the leach tank, movement vertically through one or more layers of the unsaturated zone, and then laterally through the saturated zone to a compliance location. The release model selected is the linear source term, which is based on distribution coefficients. The release model accounts for inventory changing over time with the flux released set in proportion to remaining inventory. Radionuclide transport is modeled zone by zone (each unsaturated zone sequentially, then the saturated zone) by numerically calculating flux across each zone progressively over time. In RESRAD-OFFSITE, the transport along each segment is calculated using an analytical solution of the one-dimensional transport equation that accounts for one-dimensional dispersion in the unsaturated zone and three-dimensional dispersion in the saturated zone. The solution accounts for sorption, decay, and ingrowth. Results as radionuclide concentrations are derived from a hypothetical well placed at the compliance location.

6.2 Input Parameters

Site-specific information is used when available to determine input values. Table 5 provides a selected portion of parameters values used in the RESRAD-OFFSITE model. Hydraulic properties and distribution coefficients are two important transport parameters. Site-specific information on hydraulic properties is available for the Durita site, though is not always relevant to field properties at the appropriate scale. Distribution coefficients for the different radionuclides were discussed extensively in Section 5. As discussed in Section 5, site-specific

information on distribution coefficients is not available, though geochemical information that may be helpful in estimating distribution coefficient values is estimated. Discussions on the basis and rationale for several important parameter input values are provided below, including hydraulic conductivity and effective porosity, for vertical water flux leaving the tailings tanks, contaminated zone area and thickness, and unsaturated zone thickness.

Hydraulic Conductivity and Effective Porosity

Hydraulic conductivity estimates for each segment of the transport scenario pathways utilize site-specific measurements and information. The hydraulic conductivity of the unconsolidated sediments is required for Transport Scenario Pathway 1 and 2, and of the conductive units (i.e., sand-bearing layers) of the lower Mancos Shale for Transport Scenario Pathway 2. For the former, field-based percolation tests provide an average value of 3.3e-5 cm/s (35 ft/yr) and a maximum value of 3.3e-4 cm/s (340 ft/yr). However, instead of using the percolation test results for the unconsolidated sediments, the chloride pulse event described in Section 2 is used to refine the values for both the hydraulic conductivity and effective porosity, as described below, because it represents field scale properties. For the latter, the unweathered lower Mancos, the highest values recorded from borehole tests are likely associated with the conductive layers of the Mancos Shale. The highest value recorded, 3.1e-4 cm/s (320 ft/yr), which is recorded as 98 m/yr in Table 6, is used as an input for saturated zone in Transport Scenario Pathway 2.

The hydraulic conductivity for Transport Scenario Pathway 1 is derived from the chloride pulse event expressed in data from monitoring well MW-4 in 1988 and 1989. An effective hydraulic conductivity derived from the chloride pulse that registered in MW-4 better reflects a transport parameter set that incorporates the transient nature of flow over the 10-yr period. Laboratory and field percolation tests may not reflect large-scale heterogeneity nor the periodic and localized nature of flow postulated along the bedrock-unconsolidated sediment interface. CDPHE (2016) indicated that the 22-month chloride pulse appears to correspond to the 21-month duration of the operating period of 1977 to 1979, and that it implied a 7.75-year delay (CDPHE, 2016, p. 90). Based on the information provided, the NRC staff did not find that the data showed a 7.75-year delay, and instead assumed a 10-year delay between release near the Plant Processing Building and breakthrough at the MW-4 well located 920 ft (280 m) downgradient. Rather than the 120 ft/yr (36 m/yr) rate estimated by CDPHE (2016, p.90) for the chloride, staff instead derived a transport rate of 92 ft/yr (28 m/yr). Chloride is a conservative tracer, i.e., no sorption onto solids is assumed to occur along the pathway. The chloride pulse was simulated using RESRAD-OFFSITE with a simplified form of Transport Scenario Pathway 1 to ensure that the hydrological inputs are consistent with the groundwater flow conceptualization. The input values in the RESRAD-OFFSITE model should reproduce the timing of the chloride pulse. Because of the nature of the release, particularly the source concentration, is not known, only the timing of the breakthrough is sought in the simulation. RESRAD-OFFSITE inputs for hydraulic conductivity and effective porosity can be calibrated by matching the 10-yr delay in the chloride pulse reaching well MW-4. Values of 170 m/yr (560 ft/yr) for hydraulic conductivity and 0.15 for effective porosity lead to a reasonable fit to the 10-yr delay in chloride reaching MW-4. For comparison with measured hydraulic conductivity, the

mean value from percolation test recorded in Fox (1977, 1982) is 3.34e-5 cm/s (10 m/yr) and the maximum value is 3.27E-4 cm/s (104 m/yr).

Vertical Water Flux

CDPHE (2016) reported two series of "percolation/infiltration" estimates using the HELP model (Schroeder et al., 1994). They reported estimates of 0.00043 and 0.0011 inch/yr (0.0011 and 0.0028 cm/yr) in the closure cell, and 0.00103 and 0.0019 inch/yr (0.0026 and 0.0048 cm/yr) in the leach tanks. Because no other information was provided, it is not clear if these estimates of "percolation/infiltration" reflect infiltration into the leach tanks, percolation through, or leakage out the bottom of the compacted clay layer.

A conservative estimate of the vertical water flux below the compacted clay liner is derived using the guidance in NAS (2007) that 1 to 2 percent of the precipitation in semi-arid climates will be transmitted through covers in the long-term (e.g., as the covers evolve to long-term conditions). This range is consistent with the discussion in Section 2 regarding typical infiltration and recharge rates for natural sediments in arid areas in the western U.S. that are similar to Durita. Based on a literature survey describe in Section 2.3, the leach tank cover and closure cell cover will likely evolve towards hydrologic properties similar to natural unconsolidated sediments in the area. Using the 2 percent value and conservatively assuming that the compacted clay liner does not restrict flow, the vertical flux would be 0.24 inch/yr (0.006 m/yr) for a precipitation rate of 12 inch/yr. This flux is approximately five times the flux derived from the HELP model provided in CDPHE (2016). Because the saturated hydraulic conductivity value of 0.32 m/yr (1.05 ft/yr) for the compacted clay liner in Table 5 combined with a unit hydraulic head gradient is much greater than the recharge flux rate of 0.006 m/yr (0.24 in/yr), the recharge is readily transmitted through the compacted clay liner in the steady state simulations of RESRAD-OFFSITE.

Flux through the compacted clay liner can be viewed in two ways reflecting different mechanisms for contaminants breaching the compacted clay liner. The input parameters in the RESRAD-OFFSITE model that affect the flux through a degraded liner are the liner permeability and contributing volume. The water flux carrying contaminants is assumed either to be uniform across the footprint of the leach tanks, or dominated by flux through defects in the liner. For the case of uniform flow through the compacted clay liner, water flux could be estimated using an assumed head (height) of water above the liner to determine the gradient in head and a hydraulic conductivity that reflects field measurements of liners at other sites (see Section 2.3 Site Engineered Features). Based on the literature values of laboratory core measurements used to meet design requirements compared to field measurements, a hydraulic conductivity value one order of magnitude larger than the design value reasonably reflects the field data as was discussed in Section 2. For the second case, contaminant release though defects in the compacted clay liner, a larger water flux estimate reflecting faster flow through defects could be used in combination with a smaller source amount that reflects a smaller portion of the tailings in a capture zone of the defects. In this case, the contributing volume of contaminants is more important than the hydraulic conductivity for flow through liner defects. Because defects may

occur over the entire area of the footprint of the leach tanks, the source term should be adjusted by reducing the amount of radionuclides in the contaminated volume.

An alternative approach for estimating the leakage flux below the compacted clay liner considers the hydraulic conductivity of the liner and the head difference across the layer. This approach was used in CDPHE (2016, p.85) to estimate leakage through the liner over an 8-month period. With a maximum head difference across the liner of 6 m (20 ft; the depth of the leach tank), the derived flux would be greater than the average precipitation rate when using the hydraulic conductivity in Table 5. If instead, the design criteria of liner hydraulic conductivity is used, the flux would be 62 percent of the annual precipitation rate. At some small saturated height in the leach tank, rather than the 6-m (20-ft) head difference across the liner, the derived flux would become a reasonable fraction of the precipitation rate more in line with estimated recharge in the Durita area (see Section 2.1). No basis is readily available for determining a reasonable value for the saturated thickness of tailings, therefore this approach for estimating leakage below the liner was abandoned for the long-term simulations. CDPHE (2016) stated that there were no observations of saturation in the waste tailing after operations, though the test hole temporal and spatial aspects would not be able to identify localized or evolving saturation states.

Source Term Footprint and Thickness

The combined footprint of all three leach tanks and the average waste tailings thickness are used to set the source term area and thickness in the RESRAD-OFFSITE model. Using a single area simplifies the RESRAD-OFFSITE modeling; separate simulations are required for spatially separated source terms. In addition to the area and thickness of the tailings, the inputs for RESRAD-OFFSITE for the contaminated zone also require contaminate concentrations, which are based on total curies for each radionuclide and the tonnage of waste tailings.

A rectangular area with the same area as the combined footprint of the three leach tanks is provided as input to RESRAD-OFFSITE. The footprint for each leach tank is derived from information in the Four Corners ER (1977). The rectangle is oriented such that the north-south combined dimension of LT-201 and LT-203 is parallel to the groundwater flow direction. The thickness of the tailings is based on six measurements from test boreholes reported in Hecla (1991, Volume II). The average of the measurements is 17 ft (5.2 m). An estimate of the tailings thickness is 6.2 m (20 ft) based on volume and footprint, and another estimate is 5.6 m (18 ft) based on tailings tonnage, bulk density, and footprint. Both of the estimates do not account for settling or on variations in bulk density. Therefore, the value of 5.2 m (17 ft) derived from measurements is used in the RESRAD-OFFSITE modeling.

The concentration of radionuclides in the waste tailings is estimated based on the tonnage of tailings and the activity (e.g., number of curies) estimates from Section 5.2, Contaminants Remaining In the Impoundments. The volume, and hence the choice of thickness, of the tailings does not influence the concentrations. The thickness is used in RESRAD-OFFSITE for transport calculations within the tailings, which may affect the release rate of radionuclides

below the liner if the thickness is some small value. The value of thickness used here is large enough such that the release rate, which is based on equilibration between the solid and liquid phases, is not significantly affected.

Contributing Volume

Compacted clay liners, including the liners at Durita, are designed such that only a small amount of contaminants may leak. Diffusion may be the prominent transport mechanism at the designed permeability. However, initial defects may lead to leakage at rates faster than diffusion. The style of initial defects is that of localized failure points (see discussion in Section 2). With localized failure, some fraction of the radionuclide inventory would contribute to that leaked through the defects. Two factors are considered in the selection of a contributing volume: (i) infiltration and percolation through the waste tailings is likely spatially heterogeneous, and (ii) the ratio of areas of initial and localized defects to the entire waste tailings footprint.

Spatially heterogeneous infiltration and percolation through the waste tailings likely occurs. In addition, some portion of the radionuclides are in the interior of grains and rock fragments; i.e., not all the radionuclides are in contact with the fluid phase. A consequence of this heterogeneity is that only a portion of the radionuclides would come in contact with the percolating water. The extraction efficiencies for heap leaching operation cover a wide range. At Durita, staff estimate the extraction efficiency using 5 percent sulfuric acid to be 44 percent (see Section 5). Additional applications of acid likely lead to lower and lower extraction efficiencies. In addition, the chemistry of percolating groundwater would be less efficient than the 5 percent sulfuric acid solution. Thus, the contributing volume (or fraction of radionuclides) is between 0 and 44 percent of the total activity because removal efficiencies drop with successive applications of acid.

Based on the description of the radionuclide leakage through the liner of the evaporation ponds, only small, localized areas of the liner failed (CDPHE, 2007) based on the limited lateral extent of fluid and contaminant migration found during reclamation. Though not quantified in the description, an estimate of 1 percent of the evaporation pond area with a failed liner condition is therefore conservative. There is no information for compacted clay liners in the literature, but for geomembrane liners, ranges from 1 to 20 small defects per acre are incorporated into flux calculations (e.g., LaTouche and Garrick 2012; Chapuis, 2002; NAS, 2007). The 1 percent failed liner area is assumed to be applicable to the leach tank liner. In addition, the entire volume of the waste tailings would not contribute to leaks through the 1 percent area of failed liner because any such failure would necessarily be localized to some portion of the tank. A safety factor of ten is applied to arrive at a 10 percent contributing area for the nominal case. Staff considers 10 percent to be reasonably conservative, but also performed a sensitivity analysis in Section 7.3 using 100 percent efficiency, which equates to the release of all radionuclides from the leach tanks.

Some measure of liner performance is warranted, though clearly the discussion above shows the risk of significant uncertainty in quantification of that performance. Note that the nominal case uses a compacted clay liner hydraulic conductivity value that reflects some degradation, but not total degradation of the liner, by using a hydraulic conductivity value one order of magnitude larger than the design criteria for the liner (Table 5). In Section 7 Modeling Results, therefore, the uncertainty analysis considers a contributing volume percentage that ranges from 10 to 100 percent considering dropping extraction efficiencies with successive applications of acid and efficiency of natural groundwater, and conservative estimates of liner failure based on research studies.

Layer Thicknesses

The unsaturated and saturated zones are defined differently for the two Transport Scenario Pathways.

In the conceptualization for Transport Scenario Pathway 1, a portion of the unconsolidated sediments is part of the unsaturated zone and the remainder plus a 1-m (3-ft) thickness of weathered Mancos Shale are part of the saturated zone. Unweathered Mancos Shale is not included in Transport Scenario Pathway 1. In the conceptualization for Transport Scenario Pathway 2, the entire thickness of unconsolidated sediments plus a 1-m (3-ft) thickness of weathered Mancos Shale comprise the unsaturated zone thickness. The sediments and weathered Mancos comprise Unit A in the CDPHE (2016) nomenclature (see Section 2). The unweathered portion of the Mancos Shale in the unsaturated zone is neglected because of the assumption of fast vertical flow in fractures to the conductive layers of the saturated zone lower part of the Mancos Shale.

The thickness of both the unconsolidated sediments and the weathered Mancos varies over the site. The unconsolidated sediments are thicker along the main paleochannel, which is incorporated into Transport Scenario Pathway 1, compared to other onsite locations. Use of an average thickness for the site would be conservative for performance in the RESRAD-OFFSITE model because it is likely less than the thickness of sediments along Transport Scenario Pathway 1. The average thickness of unconsolidated sediments is derived from borehole and test hole logs in Fox (1977), Hecla (1998a), and CDPHE (2015). The calculated average is 14.7 ft (4.5 m); for boreholes entirely contained in the sediments, with a listed sediment thickness of ">20 ft", a 20-ft (6-m) thickness is assigned for estimation of the average. For the other component of Unit A, the thickness of weathered Mancos Shale is not well defined. Descriptions in borehole logs based on borehole cuttings for MW-8 through MW-14 (CDPHE, 2015) indicate that the weathered Mancos Shale varies from 3 to 12 ft (0.9 to 3.7 m) thick. A thickness of 1 m (3.3 ft) is conservatively selected for the model. Therefore, a total thickness of Unit A (unconsolidated sediment plus weathered Mancos Shale) is 5.5 m (18 ft).

The saturated zone thickness of Transport Scenario Pathway 2 is the conductive layers in the lower portion of the Mancos Shale. An estimate of 3 m (10 ft) for the conductive zone beneath the site is provided in CDPHE (2016), which is consistent with the maximum thickness reflected

in the borehole data. This zone varies across the site from a single 1-ft thick (0.3-m) sandstone layer to several thin sandstone layers within a 10-ft (3-m) zone (Section 2.2).

Table 5. Parameter values used in RESRAD-OFFSITE modeling for Site Layout, Contaminated Zone, and Compacted Clay Liner; parameter values that are the same for both Transport Scenario Pathways. [1 m = 3.3 ft, 1 pCi/g = 0.037 Bq/g].

Group, Parameter	Value	Units	Comments
Meteorological Data			
Precipitation Rate	0.305	m/yr	CDPHE (2016) 12 inches/yr
Recharge	0.006	m/yr	2% of precipitation (NAS, 2007)
Contaminated Zone			
Uranium-238 concentration	112	pCi/g	See Section 5, Table 2;
Uranium-234 concentration	111	pCi/g	Values for 100% contribution
Thorium-230 concentration	870	pCi/g	
Radium-226 concentration	864	pCi/g	
Lead-210	850	pCi/g	
Bulk Density	1.27	cm^3/g	Hecla (1991), Table 3
Porosity	0.52	-	Hecla (1991), Table 3
Hydraulic Conductivity	100	m/yr	Staff judgement
Thickness	5.2	m	See text in Section 6
Compacted Clay Liner			
Thickness	0.3	m	CDPHE (2016), p.27
Hydraulic Conductivity	0.32	m/yr	See text Section 3, 1e-6 cm/s
Effective Porosity	0.1	-	Staff judgement
Bulk Density	1.86	g/cm^3	Staff judgement
Dispersivity, Longitudinal	0.03	m	Scale dependent

Table 6. Parameter values used in RESRAD-OFFSITE modeling for Site Layout, Unsaturated Zone, and Saturated Zone; parameters that may vary between Transport Scenario Pathways 1 and 2. [1 m = 3.3 ft; 1 acre = 4050 m^2]

Group, Parameter	Scenario 1	Scenario 2	Units	Comments
Site Layout				
Leach Tank to Site boundary	494	494	m	Measured on Google Maps
Source Area Footprint	22.2	22.2	acres	Combined, all three tanks
Length Parallel to Flow	247	247	m	See text in Section 6
Length Perpendicular to Flow	364	364	m	
Unsaturated Zone Below				
Thickness	2.5	5.5	m	See text Section 6
Hydraulic Conductivity	170	170	m/yr	See text
Effective Porosity	0.15	0.15	-	See text
Bulk Density	1.8	1.8	g/cm^3	Hecla 1991 Table 3
Dispersivity, Longitudinal	0.4	0.4	m	Scale dependent
Saturated Zone				
Aquifer thickness	3	3	m	Unit A; Unit C
Hydraulic Conductivity	170	98	m/yr	See text

Hydraulic Gradient	0.033	0.033	-	CDPHE (2016)
Effective Porosity	0.15	0.2	-	Staff judgement
Bulk Density	1.8	1.8	g/cm^3	Hecla (1991) Table 3
Dispersivity, Longitudinal	5	5	m	Scale dependent
Dispersivity, Transverse	0.5; 0.05	0.5; 0.05	m	Horizontal; Vertical
Distribution Coefficient				
Uranium K_d Contaminated Zone	15	15	L/kg	Oxic, high SO4^{2-}, low pH
Uranium K_d Unsaturated Zone	15	15	L/kg	Oxic environment, Section 5
Uranium K_d Saturated Zone	15	100	L/kg	Oxidizing; Reducing
Thorium K_d	1700	1700	L/kg	See Section 5, Table 3
Radium K_d	70	70	L/kg	See Section 5, Table 3
Lead K_d	100	100	L/kg	See Section 5, Table 4

7   Modeling Results

The approach taken for simulating the release of contaminants from the leach tanks is to use conservative assumptions and input values that are described in Section 4, 5, and 6. The results using this approach are referred to as the nominal case results for each transport scenario pathway. The nominal case is not the expected condition, but rather a conservative measure used to judge safety. The objective is to ascertain how soon after the 1,000-yr performance period significant contamination might occur at the compliance location using the conservative assumptions of the nominal case. After completing the nominal case simulations, sensitivity analyses are performed for several of the most important uncertain parameter inputs. The sensitivity analyses illustrate the effect on performance that those parameters exhibit, particularly for the timing of peak concentration in regard to the 1,000-yr performance period.

The primary benchmarks in evaluating simulation results are time to peak concentration and peak concentration. Also considered are the EPA drinking water standards for total uranium (30 μg/L) and radium-226+radium-228 [5 pCi/L (0.19 Bq/L)] that provide the context for evaluating simulation results. The vanadium benchmark of 50 μg/L is taken from the notification level for drinking water in the State of California. In considering these benchmarks, staff notes that no contamination of the groundwater had been found during the period of monitoring from 1978 to 1998 [though the monitoring well placement may not be adequate (NRC, 2014)], and regional measurements of background groundwater in the Mancos Shale exhibit a range of uranium and radium-226 concentrations that may exceed the drinking water standards absent any contamination from Durita or any other uranium milling site. In addition, simulation results presented in the tables carry significant figures beyond the level justified by the modeling inputs and assumptions; the significant figures in the table are retained to facilitate traceability with modeling output files and illustrate consistency between cases only.

The following subsections contain a description of the evolution of the contaminated zone (leach tanks), the conservative case results, and sensitivity analyses for important uncertain inputs.

7.1   Evolution of Contamination in Leach Tank

Figure 5 illustrates the calculated evolution of uranium-238, uranium-234, thorium-230, radium-226, and lead-210 in the waste tailings. The drop in the concentration over time is caused both by the inventory-tracked release from the contaminated zone and by decay. The release model selected for use in RESRAD-OFFSITE is based on distribution coefficients partitioning the radionuclides between the solid and water phases. Thorium-230, radium-226, and lead-210 remain in secular equilibrium and comprise the upper set of curves, which initially represent the amount of activity in the ore. The extraction of uranium during processing, both at Naturita and Durita, cause the uranium isotopes curves to be offset and lower than the thorium-230, radium-226, and lead-210 set of curves.

Uranium remaining in the waste tailings supports the continued production of its daughter products. If uranium was not present in the waste tailings after operations were complete, the shorter half-lives of thorium-230 (half-life 75,400 years), radium-226 (1,600 years), and lead-210 (22.3 years) (NRC, 1996) would cause those daughter products in the source term to drop off more sharply than shown in Figure 5.

Figure 5. Evolution of contamination in the waste tailings source area. [1 pCi/L = 0.037 Bq/g].
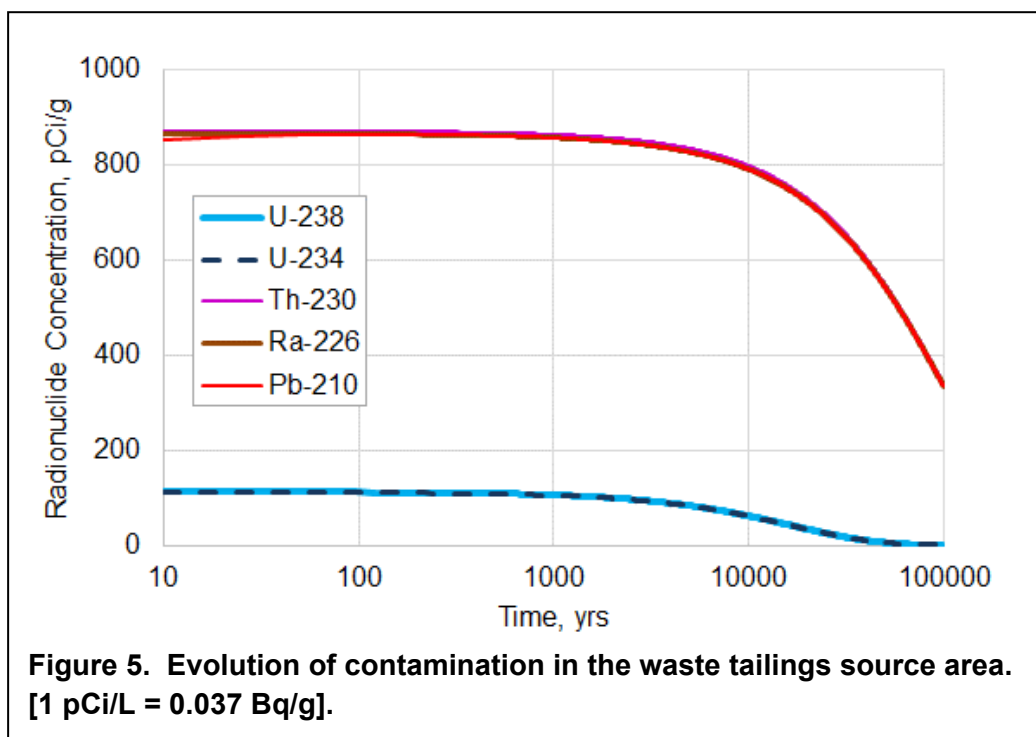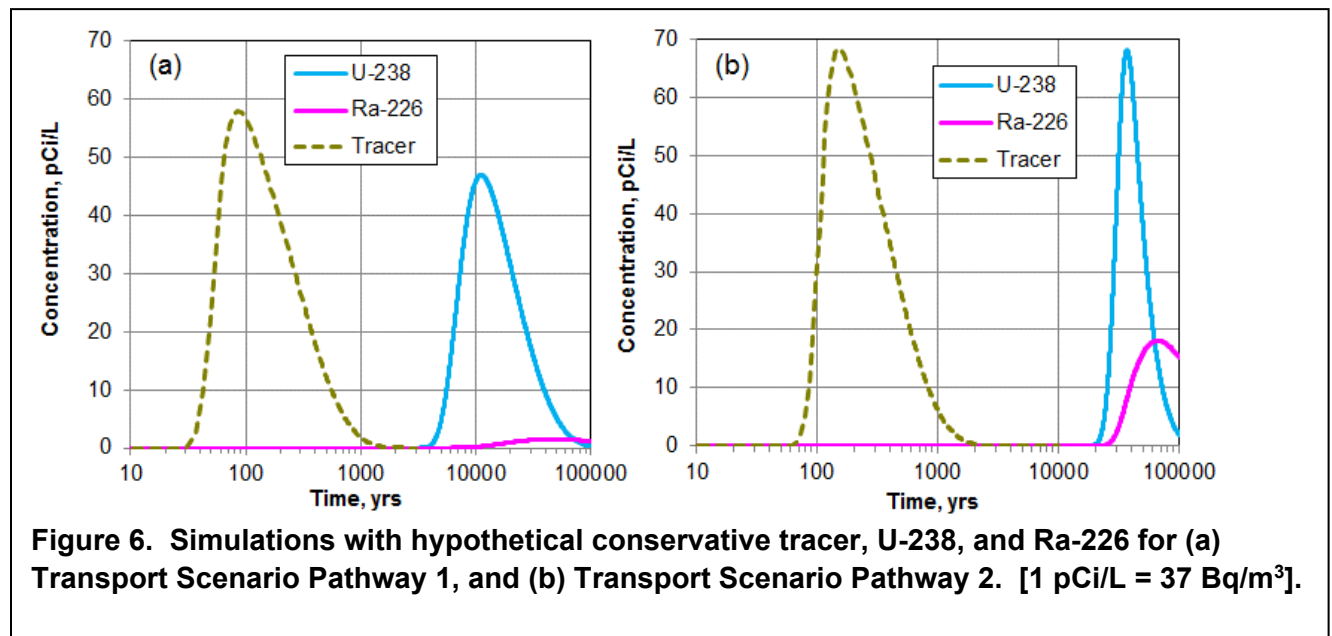
7.2   Characteristic of Transport Pathway

Simulating a conservative tracer provides a measure of the time length scale of a transport pathway. A conservative tracer is a contaminant with a distribution coefficient equal to zero, which means that contaminant remains in the water phase and does not sorb onto solids or precipitate as minerals. Retardation is 1 for a contaminant with a K_d of 0 L/kg. Retardation

reflects the difference between the travel time of a water parcel based on the linear velocity of the water and the time that a sorbing contaminant would take to migrate the same distance. The higher the retardation factor, the slower the transport. The retardation factor is proportional to the K_d and bulk density and inversely proportional to the effective porosity.

For uranium, with a K_d=15 L/kg, and effective porosity and bulk density values for the unconsolidated sediments provided in Table 5, the retardation factor would be 181. This means that uranium would take approximately 181 times as long to reach the compliance location as a conservative tracer. Radionuclides with different distribution coefficients will have different retardation factors for the same pathway. With a K_d of 70 L/kg, the retardation coefficient for radium for the unconsolidated sediments would be 841. For a multi-segment pathway with different properties for each segment, the calculation is more complicated; RESRAD-OFFSITE handles that complexity.

The hypothetical, conservative tracer curves (dashed lines) in Figures 5a and 5b illustrate the characteristic time profile for each transport scenario pathway. The travel times for a conservative tracer for Transport Scenario Pathways 1 and 2 are approximately 90 years and 150 years, respectively. These transport times reflect the effect of contaminant migration in the multi-segment pathway. Each segment has different properties; the segments are the waste tailings in the leach tank (as part of the release model), compacted clay liner, unsaturated zone, and saturated zone. Based on peak concentrations and the conservative tracer for Transport Scenario Pathway 1, a contaminant with a K_d less than 1 L/kg would lead to a retardation factor such that contaminant peak concentrations would reach the compliance location within 1,000 years. For Transport Scenario Pathway 2, the K_d would need to be approximately 0.6 L/kg for the peak to reach the compliance location within 1,000 years. As discussed in Section 5.3, Behavior of Radionuclides in Groundwater System and shown in Table 4, none of the distribution coefficients for radionuclides of concern can reasonably be expected to be this small for Durita.

Figure 6. Simulations with hypothetical conservative tracer, U-238, and Ra-226 for (a) Transport Scenario Pathway 1, and (b) Transport Scenario Pathway 2. [1 pCi/L = 37 Bq/m^3].

7.3   Results for Transport Scenario Pathway 1

Transport Scenario Pathway 1 results are shown in Figure 7 and tabulated in Table 7. There are co-incident peaks for uranium-238 and uranium-234 at 10,600 years and separate peaks for radium-226 and lead-210 beyond 46,200 years. The pulse of uranium isotopes reaches the designated uppermost saturated layer first among the constituents because of the small value for the uranium distribution coefficient. With the large K_d value selected for thorium, the small amount shown in Figure 7 is mostly due to ingrowth from uranium decay along the multiple segments of the transport pathway. Some portion of the radium-226 and lead-210 is also likely due to ingrowth, which is supported by the close timing of their peak concentrations even though they have different distribution coefficient values. A separate simulation performed without uranium isotopes confirmed that ingrowth from the uranium-238 decay chain is principally responsible for the radium-226 and lead-210 peaks shown in Figure 7. Therefore, the amount of uranium in the waste tailings and its small distribution coefficient are the most important factor for Transport Scenario Pathway 1 because it drives the timing and peak concentration at the compliance location.

Whereas the peak concentration at the compliance location for the uranium isotopes occurs at 10,600 years (Table 7), the EPA drinking water standard of 30 μg/L total uranium is reached at 5,044 years. Peak total uranium at the compliance location is 138 μg/L, which is above the EPA drinking water standard of 30 μg/L. Because of the respective specific activities of the uranium isotopes, uranium-234 does not contribute much to the mass-based concentration for total uranium. A peak concentration of 46.9 pCi/L (1,735 Bq/m^3) for uranium-238 at the compliance location is nearly 1 percent of the groundwater concentration in the source area, assuming sorption equilibrium between the tailings solids and the pore water in the leach tanks. For radium-226, the peak concentration of 1.5 pCi/L (55 Bq/m^3) occurs after 46,230 years (Table 7), and is below the EPA drinking water standard of 5 pCi/L (185 Bq/m^3) for radium-226 even at that time.

Vanadium concentration is 0 at 1,000 years. The benchmark from Table 4 of 50 μg/L, which is the notification level for drinking water for the State of California, is reached in the simulation at approximately 18,230 yrs, and peak value is not reached until approximately 36,925 yrs.

For Transport Scenario Pathway 1, contaminated groundwater in the unconsolidated sediments would not reach the designated uppermost saturated layer at the site boundary unless the primary assumption of this pathway was violated. Inherent in this pathway is the absence of fracture flow vertically through the Mancos Shale until the known fault along Dry Creek. A simulation is performed with the well placed at the fault at Dry Creek instead of at the site boundary. The transport distance is 730 m (2,400 ft) instead of 494 m (1,620 ft). The simulation results indicate that the peak concentrations for uranium-238 and uranium-234 occur at approximately 11,100 years, which is 500 years later than occurs for a well at the site boundary. The peak concentration below the site boundary is not significantly smaller than that at Dry Creek. The EPA drinking water standard of 30 μg/L total uranium is also reached approximately 500 years later at the Dry Creek fault compared to that below the site boundary.
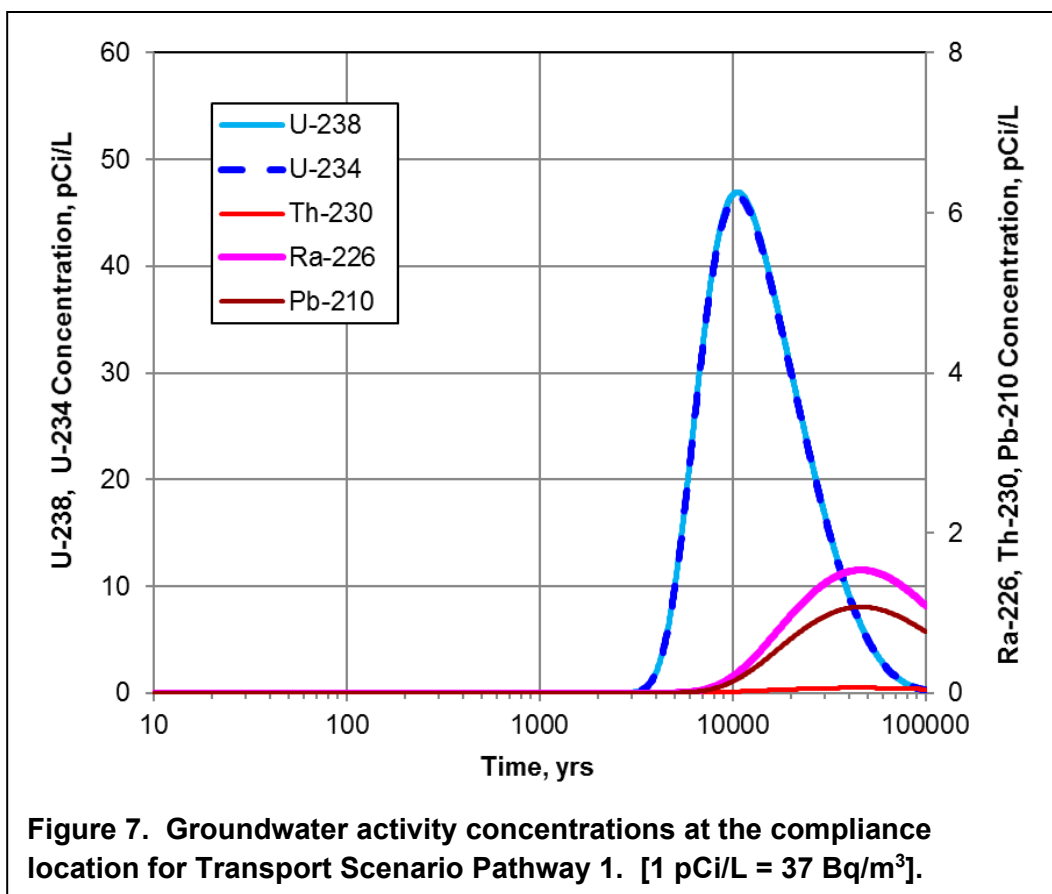
Figure 7. Groundwater activity concentrations at the compliance location for Transport Scenario Pathway 1. [1 pCi/L = 37 Bq/m³].

Table 7. Simulation results for nominal case for Transport Scenario Pathways 1 and 2. [1 pCi/L = 37 Bq/m³].

Radionuclide	Transport Scenario Pathway 1 At Dry Creek		Transport Scenario Pathway 2 At Site Boundary	
	Peak Concentration, pCi/L	Time to Peak, years	Peak Concentration, pCi/L	Time to Peak, years
Uranium-238	46.9	10,600	68.1	36,900
Uranium-234	46.6	10,600	67.6	36,900
Thorium-230	0.06	44,270	0.75	67,600
Radium-226	1.5	46,230	18.1	66,500
Lead-210	1.1	46,205	12.7	66,600

7.4   Results for Transport Scenario Pathway 2

Transport Scenario Pathway 2 simulation results are shown in Figure 8 and Table 7. Again, uranium-238 and uranium-234 are the primary concerns for this pathway. The peak concentration of uranium-238 is 68 pCi/L (2,520 Bq/m³) and the total uranium benchmark of 30 μg/L is reached at approximately 25,400 years. The highest radium-226 activity concentration

reaches approximately 18 pCi/L (666 Bq/m³) and the EPA drinking water standard of 5 pCi/L (185 Bq/m³) is reached after 33,400 years.

Vanadium concentration is 0 at 1,000 years. The benchmark from Table 4 of 50 µg/L, which is the notification level for drinking water for the State of California, is reached in the simulation at approximately 30,820 yrs, and peak value is not reached until approximately 54,600 yrs.

Figure 8. Groundwater activity concentrations at compliance location for Transport Scenario Pathway 2. [1 pCi/L = 37 Bq/m³].
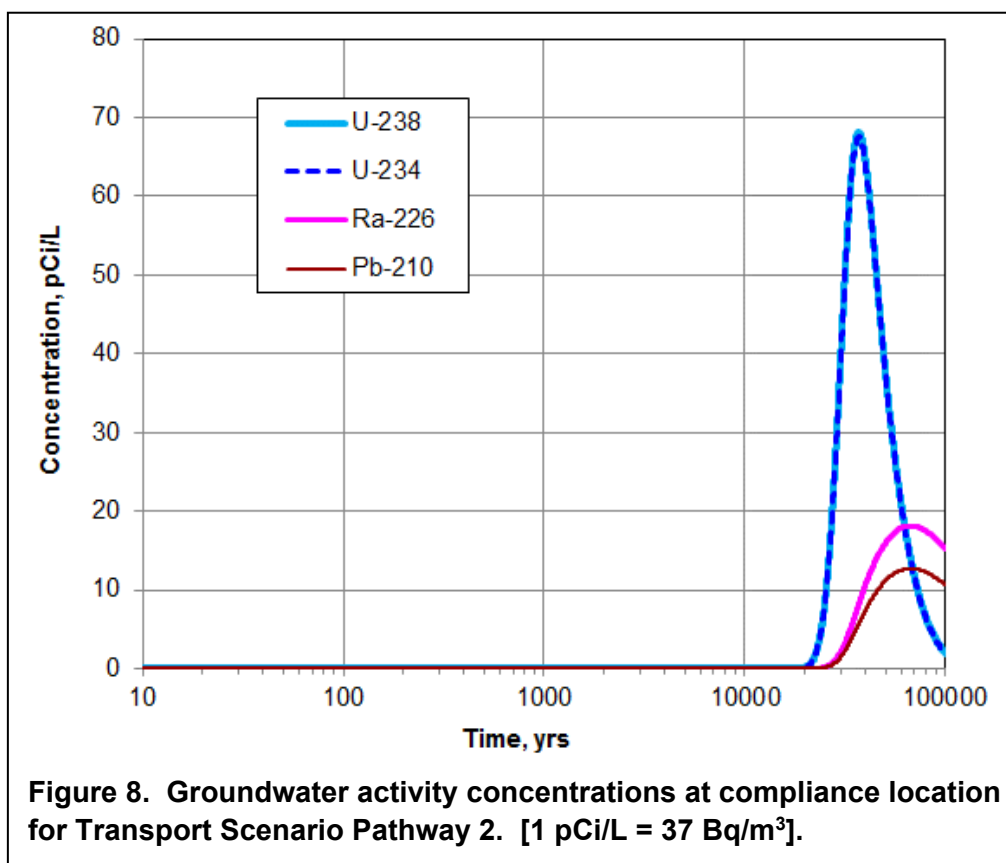
7.5   Sensitivity Analysis for Uncertainty

The nominal case described above is substantially conservative, such that the results should not be considered expected contamination reaching the compliance location. However, it is instructive to ascertain the performance consequences of further perturbing important inputs. The results of RESRAD-OFFSITE simulations are described in this subsection for three inputs that are both important to performance and uncertain because of a reliance on indirect site-specific information. Other uncertain inputs have already been incorporated into the model, such as the presence or absence of vertical fractures in the Mancos Shale, or have been determined not to be important to performance using one-off sensitivity analyses in RESRAD-OFFSITE. Results from sensitivity analyses for three inputs are presented below. These three inputs were selected because results are highly sensitive to input values, and because there is

little site-specific supporting data for the inputs. The three inputs analyzed below are the uranium K_d, leakage flux below the leach tanks, and volume of leach tank contributing to leaks.

Distribution Coefficient for Uranium

Data in Table 4 indicate that a generic value of the uranium distribution coefficient from published surveys of measurements from other sites would be 50 L/kg or greater. However, several sites in the western U.S. exhibited smaller values of K_d for uranium (see Section 5, Distribution Coefficient for Uranium). To illustrate the effect of uranium K_d values on concentrations at the compliance location, simulations were performed with K_d values of 15 (nominal case), 10, and 5 L/kg. Results of the simulations in Table 8 show that year to peak concentration, peak concentration, and time to total uranium concentration of 30 μg/L all change with values of the distribution coefficient. Changing from the highest uranium K_d to the lowest results in nearly a 7,000-year decrease in the time to peak concentration and an increase in the peak concentration from approximately 47 to 137 pCi/L (1,740 to 5,070 Bq/m^3) uranium-238. Uranium still does not reach the compliance location by 1,000 years, even with the smallest uranium K_d value, however.

Leakage Flux Exiting Leach Tanks

Simulations were performed with parameter input values that lead to leakage rates of 0.012, 0.006, and 0.003 m/yr (0.04, 0.02, and 0.01 ft/yr) exiting the compacted clay liner. The flux value for the nominal case is 0.006 m/y (0.02 ft/yr), which is 2 percent of annual precipitation. The RESRAD-OFFSITE results in Table 8 show that the peak concentration scales with flux rate, and the timing of the peak shifts to earlier times as the flux increases. The change in concentration between the three simulations is likely caused by the amount of radionuclides derived from the release model in RESRAD-OFFSITE. Even with the highest leakage flux rate, the 6,032-year time to peak at the compliance location remains well beyond the 1,000-yr performance period.

Volume of Leach Tank Contributing to Leak

A conservative input for the volume of the leach tanks contributing to a release would be that radionuclides in the entire volume are releasable through a uniformly and completely degraded compacted clay liner. The nominal case used a compacted clay liner with 10 percent of the contaminants releasable. To illustrate the effect of volume contributing to the release, percentages of 100, 50 and 10 percent are simulated. Logically, the concentration results at the compliance location should scale to the source volume fraction, and the timing of the peak concentration should not change.

Simulation results confirm no significant change in the timing of the peak concentration, but the expected large change occurs in the concentration value of the peak. As with the uranium K_d and the flux rate, this sensitivity analysis is pertinent to Transport Scenario Pathway 1 only. Results of the nominal case for Transport Scenario Pathway 2 would not be expected to change

significantly in regard contaminant levels approaching the 1,000-yr performance period because the breakthrough curves are far beyond 1,000 years (see for example, Figure 6). Peak concentration for Transport Scenario Pathway 1 reaches nearly 470 pCi/L (17,400 Bq/m³) uranium-238 for 100 percent contributing source volume, 235 pCi/L (8,700 Bq/m³) uranium-238 for 50 percent contributing source volume, and 47 pCi/L (1,740 Bq/m³) uranium-238 for 10 percent contributing source volume. The timing of the peak concentration does not change, however, the time when total uranium reaches 30 µg/L changes from 3,077 to 4,037 to 5,044 years for 100, 50, and 10 percent contributing source, respectively.

Table 8. Sensitivity to uranium distribution coefficient (K_d) for the unconsolidated sediments and weathered Mancos (Hydrological Unit A), and flux through compacted clay liner for Transport Scenario Pathway 1. [1 pCi/L = 37 Bq/m³, 1 m/yr = 3.3 ft].

Uncertain Parameter	Uranium-238 pCi/L Concentration at 1,000 years	Year When Total U Reaches 30µg/L	Peak Concentration Uranium-238 pCi/L	Year of Peak
Uranium Distribution Coefficient, L/kg				
K_d=15 L/kg, Base Value	0	5,044	46.9	10,600
K_d=10 L/kg	0	3,155	69.9	7,107
K_d= 5 L/kg	0	1,454	137.2	3,590
Flux Through Clay Liner				
0.012 m/yr	0	2,936	93.6	6,032
0.006 m/yr, Base Value	0	5,044	46.9	10,599
0.003 m/yr	0	9,994	23.7	19,037

8   Summary and Conclusions

This report evaluates the designated uppermost aquifer and estimates the consequence of a potential release of radionuclides from the Durita impoundments to the groundwater system and migration offsite.

The designated uppermost aquifer in the CRR is the lower Mancos Shale conductive horizon, which includes hydraulically connected thin sand layers, and the alternating shale and sand layers at the transition between the Mancos Shale and the upper portion of the underlying Dakota Sandstone. In the description of this horizon, Hecla (1992) indicated that the uppermost aquifer is not a potential water resource because (i) the quantity and yield rate are too low, (ii) the areal extent is limited, and (iii) the water quality is poor. Based on site data, the NRC staff agrees with this description of the saturated horizons near the base of the Mancos Shale at Durita, but accepts this designation for the purpose of this analysis because it is not "interconnected" (hydraulically connected) with natural aquifers lower in the system and therefore a conservative designation than a deeper, potentially more productive aquifer. The staff notes that the nearest well to the Durita site pumps water from an aquifer is the lower Dakota Sandstone. This well, the Coke Oven Ranch well, is not at risk for contamination from

the Durita site based on geochemical stratification of saturated horizons below the site, the confined nature of the lower aquifers, and groundwater flow directions that are constrained by the structural geology of the area.

For this groundwater evaluation, a conceptual model was developed that indicated two possible transport scenario pathways for consequence modeling of a release radionuclides from the impoundments at Durita. CDPHE and the NRC staff have both cited features at the site that may reduce the likelihood of releases or the consequences if a release of radionuclides occurs from the leach tanks or closure cell (e.g., NRC, 2007; CDPHE, 2005a). These features include low precipitation and recharge rates, compacted clay layers in the tailings cover and liner, low permeability shale layers below the site, and geochemical conditions that would inhibit the rapid migration of radionuclides. These features and conditions were analyzed in terms of available site-specific information and uncertainty as part of the development of conceptual models, and incorporated as appropriate into input parameters for two transport pathways. These transport pathways incorporated readily supportable features, and assigned conservative estimates for inputs with only indirect information from the site.

Previously, CDPHE (2005a) and NRC (2007) modeled the consequence of potential releases using travel time calculations. The travel time calculations incorporated (i) particle velocity estimates based on groundwater gradient, hydraulic conductivity, and effective porosity estimates, (ii) instantaneous releases of a contaminant, (iii) retardation due to sorption, and (iv) distance to the compliance location. The CDPHE (2005a) travel time estimate was 22 million years, with a safety factor adjustment to 22,000 years. Using ranges for sorption, permeability, and distance to compliance location, NRC (2007) found some combinations of the uncertain parameters led to travel times of less than 1,000 years. Because some combinations of parameters led to travel times less than the performance period, more detailed analysis of the site's features was necessary.

The consequences of a hypothetical releases are quantified using flow and transport modeling that incorporates (i) an updated conceptual model for groundwater flow at the site, (ii) an estimate of concentrations of radionuclides in the impoundments, and (iii) expected geochemical characteristics for the site. The calculations in this report include a linear release rate from a source volume, radionuclide decay and ingrowth, and transport through the multiple geologic layers that considers geochemical conditions and radionuclide sorption for each layer. In addition, concentrations of uranium-238, uranium-234, thorium-230, radium-226, lead-210, and vanadium in the impoundments (leach tanks and closure cell) are estimated based on analysis of geochemical conditions in the ore, and historical processing efficiencies at Naturita and Durita. The quantity of radium-226 curies listed on the monument at the site is a factor of two smaller than that from the NRC staff estimate. The flow and transport modeling in this report conservatively uses the NRC staff estimate for all radionuclides in the leach tanks. Because site-specific information was not available for some parameters, staff uses inferences and indirect methods to develop the scenarios and inputs needed for modeling. The lack of site-specific information led to the use of conservative choices for some inputs and guides the sensitivity analyses for important uncertain inputs. A nominal case is generated where some

inputs use site-specific information, and other inputs based on indirect site information use conservative input values.

Uranium is the primary concern for Transport Scenario Pathway 1, which likely has an oxidizing environment in the unconsolidated sediments and weathered Mancos Shale. Simulation results with conservative, but reasonable, values of inputs indicate that peak concentrations are reached at approximately 10,600 years. Total uranium of 30 μg/L, which is an EPA primary drinking water standard, is reached after 5,044 years. The peak concentration of 1.5 pCi/L (55 Bq/m^3) radium-226 occurs after 46,230 years, and is below the EPA drinking water standard of 5 pCi/L (185 Bq/m^3) for radium-226. Sensitivity analyses were performed for several uncertain parameters that had little or indirect site-specific supporting information. These sensitivity analyses do not reflect likely conditions at the Durita site, but rather test the effects of perturbing important parameters on estimates of radionuclide concentration at the compliance location. Using time of peak concentration, results of the most conservative input perturbations are approximately (i) 3,590 years for uranium distribution coefficient of 5 L/kg, (ii) 6,032 years for flux through the leach tank clay liner of 0.012 m/year (0.48 in/yr), or 4 percent of precipitation, and (iii) 4,037 years for 100 percent contributing volume.

For Transport Scenario Pathway 2, uranium and radium-226 reach the compliance location further out in timeline compared to Transport Scenario Pathway 1. For Pathway 2, contaminants are assumed to migrate instantaneously through vertical through-going fractures in the unweathered bedrock to conductive sand layers in the lower Mancos Shale. In the reducing environment of the lower Mancos Shale, uranium is much less mobile than in the oxidizing environment of near-surface unconsolidated sediments. Results with conservative, but reasonable values of inputs indicate that the peak uranium-238 concentration of 68 pCi/L (2,520 Bq/m^3) is reached after 36,900 years, and the EPA drinking water standard of 30 μg/L is reached after 25,400 years. The peak radium-226 concentration of 18 pCi/L (666 Bq/m^3) is reached after 66,500 years, and the EPA drinking water standard of 5 pCi/L (185 Bq/m^3) is reached after 33,400 years. No sensitivity analyses are presented in this report for this transport scenario pathway because the peak concentrations are well beyond 1,000 years.

None of the analyses using conservative inputs indicate that (i) time to peak concentrations, or (ii) time when total uranium and radium-226 concentrations reach EPA drinking water standards approach the 1,000-yr performance period. In addition, the two transport scenario pathways are end member conceptualizations. Releases from the leach tanks may partition between the two pathways. If so, the resulting peak concentrations would lie between the two end member results because the characteristic travel times of the two transport scenario pathways differ. Similarly, the time at peak concentration would either match one of the individual pathway times, or lie between the two end member results.

Numerous conservative assumptions are made for these analyses. Several of the most important conservative assumptions are:
- The NRC staff derived contaminant concentrations in the leach tank are used in the modeling instead of the concentration indicated by the site monument; the derived

concentration of radium-226 is approximately twice that listed on the monument; uranium, thorium, and lead consistent with the NRC staff estimate for radium-226 are used in the simulations as well

- The engineered features are neglected (compacted clay layer in the leach tank cover) or assumed to be degraded (compacted clay layer in the leach tank liner)
- Leakage below the leach tanks is assumed to be continuous at a flux rate that assumes initial defects and a uniformly degraded liner; leakage during operations would likely be greater than after operations, particularly since the tanks were drained.

In summary, a conservative scenario analysis that includes a combination of several worst case assumptions, conservative inputs, and sensitivity analyses for several important parameters representing site features indicates that the Durita site would not likely pose a risk to the public health and safety or environment over the 1,000-yr performance period. Furthermore, the closest aquifer capable of producing usable water does not appear to be in hydraulic connection with the compliance-designated uppermost saturated layers at the Durita site.

9 References

AK GeoConsult, Inc. 1993. Geochemical Characterization of Raffinate and Shale from the Durita Uranium-Raffinate Disposal Site. Albuquerque, New Mexico. ADAMS Accession Number ML14300A042.

Amayri, S., D.R. Frohlich, U. Kaplan, N. Trautmann, and T. Reich. 2016. Distribution coefficients for the sorption of Th, U, Np, Pu, and Am on Opalinus Clay. Radiochimica Acta 104(1):33-40.

Bedinger, M.S. 1987. Summary of Infiltration Rates in Arid and Semiarid regions of the world, with an annotated bibliography. U.S.G.S. Open-File Report 87-43. U.S. Geological Survey, Denver, Colorado.

Benson, C.H., W.H. Albright, D.O. Fratta, J.M. Tinjum, E. Kucukkirca, S.H. Lee, J. Scalia, P.D. Schlicht, and X. Wang. 2011. Engineered Covers for Waste Containment: Changes in Engineering Properties and Implications for Long-Term Performance Assessment. NUREG/CR-7028. ADAMS Accession Number ML12005A110. U.S. Nuclear Regulatory Commission.

Benson, C.H. 2001. Waste Containment: Strategies & Performance. Australian Geomechanics 36(4):1-24.

Benson, C.H., D.E. Daniel, and G.P. Boutwell. 1999. Field Performance of Compacted Clay Liners. Journal of Geotechnical and Geoenvironmental Engineers, Volume 125(5), p.390-403.

CDPHE. 2016. Final Completion Review Report for the Hecla Mining Company Durita Facility Montrose County, Colorado. Colorado Radioactive Materials License No. 317-02. Colorado Department of Public Health and Environment. ADAMS Accession Number ML17243A414.

CDPHE. 2015. Durita Boring Logs MW-1 through MW-14. Provided to NRC by Colorado Department of Public Health and Environment on or before 1/22/2015. ML15022A574.

CDPHE. 2007. Colorado Radioactive Materials License #317-02, Response to NRC Geologists Report Regarding Groundwater at the Hecla-Durita Site. Letter from J. Vranka (CDPHE) to J. Schlueter (NRC) November 8, 2007. Colorado Department of Public Health and Environment. ADAMS Accession Number ML073190102 or ML17243A360.

CDPHE. 2005a. Draft Completion Review Report for the Hecla Mining Company Durita Site. Colorado Department of Public Health and Environment. ADAMS Accession Number ML053570032 or ML17243A185.

CDPHE. 2005b. Facsimile from CDPHE to T. Brock (NRC) September 7, 2005, with attachments: (i) a December 15, 1997, letter from Monster Engineering to CDPHE, (ii) pages 9 and 10 of "Hecla Mining Company - Durita Site Closure Cell Cover Construction Compliance with Criterion 6 March 26, 1999", (iii) a table titled 1997 Closure Cell Soil Layer Conditions, a description of Modification 22, and (iv) two figures with cross-sections of the Closure Cell. ADAMS Accession Number ML053560292.

CDPHE. 2005c. Email from T. Brock (CDPHE) to Bill VonTill (NRC) April 20, 2005, with attachments: (i) 1992 Hecla letter to CDPHE on the groundwater detection monitoring program, (ii) partial copy of comments and responses on the detection monitoring program, (iii) Hecla groundwater travel time calculations, and (iv) correspondence from CDPHE to Hecla on long-term monitoring program. ADAMS Accession Number ML14055A291.

CDPHE. 1998. Request for Elimination of Ground Water Monitoring Colorado Radioactive Material License (CRML) No. 317-02. Letter (partial) and attachment from Colorado Department of Public Health and Environment to Hecla (G. Gamble) December 28, 1998. Facsimile sent by CDPHE (P. Stoffey) to NRC (D. Sollenberger) on April 12, 2006

Curtis, G.P., P. Fox, M. Kohler, and J.A. Davis. 2004. Comparison of in situ uranium K_D values with a laboratory determined surface complexation model. Applied Geochemistry 19 (2004) 1643–1653.

Dames & Moore. 1978. Analysis of Tailings Disposal for Ranchers Exploration and Development Corporation Naturita, Colorado. Supplement to the Source Material License Amendment Application. RAD-41 317-02, Revision 2. ADAMS Accession Number ML14300A130.

Davis, J.A., and G.P. Curtis. 2003. Application of Surface Complexation Modeling to Describe Uranium (VI) Adsorption and Retardation at the Uranium Mill Tailings Site at Naturita, Colorado. NUREG/CR-6820. ADAMS Accession Number ML040210168. U.S. Nuclear Regulatory Commission.

DOE. 2011. Natural Contamination from the Mancos Shale. U.S. Department of Energy Report LMS/S07480, ESL-RPT-2011-01.

DOE. 2005. Office of Legacy Management Long-Term Surveillance Plan for the Durita (UMTRCA Title II) Disposal Site Montrose County, Colorado. Draft Final January 2005. DOE-LM/GJ765-2004. U.S. Department of Energy Office of Legacy Management, Grand Junction, Colorado. ADAMS Accession Number ML050410364 or ML17243A178.

DOE. 2003. Environmental Assessment of Ground Water Compliance at the Naturita, Colorado, UMTRA Project Site. DOE/EA-1466. U.S. Department of Energy, Grand Junction, Colorado.

DOE. 1982. Summary History of Domestic Uranium Procurement Under U.S. Atomic Energy Commission Contracts Final Report. Report GJBX-220(82), H. Albrethsen Jr. and F.E. McGinley, October 1982. U.S. Department of Energy, Grand Junction Area Office, Colorado. https://dspace.library.colostate.edu/handle/11166/85577?show=full; accessed August 30, 2017.

DOE. 1981. Naturita Site, Naturita, Colorado, Engineering Assessment of Inactive Uranium Mill Tailings. Report DOE/UMT-012, FBDU 360-08, July 1981. Prepared by Ford, Bacon & Davis Utah, Inc. for U.S. Department of Energy Albuquerque Operations Office, Albuquerque, New Mexico. ADAMS Accession Number ML993070180.

DOE. 1977. Phase II – Title I, Engineering Assessment of Inactive Uranium Mill Tailings, Naturita Site, Naturita, Colorado. Report GJT-8S, FBDU 130-07. Prepared by Ford, Bacon & Davis Utah, Inc. for U.S. Department of Energy Albuquerque Operations Office, Albuquerque, New Mexico. https://www.osti.gov/scitech/servlets/purl/5454802; accessed August 29, 2017.

EPA. 2005. Partition Coefficients for Metals in Surface Water, Soil, and Waste. EPA/600/R-05/074. U.S. Environmental Protection Agency, Office of Research and Development, Washington, District of Columbia.

EPA. 2004. Understanding Variation in Partition Coefficient, K_d, Values, Volume III: Review of Geochemistry and Available K_d Values for Americium, Arsenic, Curium, Iodine, Neptunium, Radium, and Technetium. EPA 402-R-04-002C. U.S. Environmental Protection Agency

EPA. 2002. Assessment and Recommendations for Improving the Performance of Waste Containment Systems. EPA/600/R-02/099. U.S. Environmental Protection Agency, Office of Research and Development, Cincinnati, Ohio.

EPA. 1999a. Understanding Variation in Partition Coefficient, K_d, Values, Volume II: Review of Geochemistry and Available K_d Values for Cadmium, Cesium, Chromium, Lead, Plutonium, Radon, Strontium, Thorium, Tritium (3H), and Uranium. EPA 402-R-99-004B. U.S. Environmental Protection Agency, Office of Air and Radiation.

EPA. 1999b. Understanding Variation in Partition Coefficient, K_d, Values, Volume I: The K_d Model, Methods of Measurement, and Application of Chemical Reaction Codes. EPA 402-R-99-004A. U.S. Environmental Protection Agency, Office of Air and Radiation.

Four Corners ER. 1977. Proposed Uranium Mill Tailings Leaching Operation at Naturita, Colorado, Environmental Report Ranchers Exploration and Development. Four Corners Environmental Research, Durango, Colorado. ADAMS accession number ML14300A118.

Fox. 1982. Preliminary Geological, Hydrological and Geotechnical Evaluation of the Coke Oven Site, Montrose County, Colorado. F.M. Fox & Associates, Inc., Wheat Ridge, Colorado. ADAMS Accession Number ML14300A126.

Fox, et al. 2006. The effect of calcium on aqueous uranium (VI) speciation and adsorption to ferrihydrite and quartz. Geochimica et Cosmochimica Acta 70, 1379–1387.

Gamerdinger, A.P., D.I. Kaplan, D.M. Wellman, and R.J. Serne. 2001a. Two-region flow and decreased sorption of uranium (VI) during transport in Hanford groundwater and unsaturated sands. Water Resources Research 37(12):3155-3162.

Gamerdinger, A.P., D.I. Kaplan, D.M. Wellman, and R.J. Serne. 2001a. Two-region flow and rate-limited sorption of uranium (VI) during transport in an unsaturated silt loam. Water Resources Research 37(12):3147-3153.

Glader. 2007. Durita License Termination - NRC Review of Groundwater Evaluation/ Completion Review Report. Hecla letter (Glader) to Sollenberger (NRC) (Figure 1 modified from this letter and attachment). ADAMS Accession Number ML071290608.

Hecla. 2007. Durita License Termination – NRC Review of Groundwater Evaluation/Completion Review Report. Letter from P.L. Glader (Hecla) to D. Sollenberger (NRC) May 7, 2007. ADAMS Accession Number ML071290608.

Hecla. 1998a. 1998 Annual Ground Water Monitoring Report. Hecla Mining Company Durita Site. ADAMS Accession Number ML14070A495.

Hecla. 1998b. 1998 Durita Site Reclamation and Construction Verification Report. Prepared by Monster Engineering, Inc., September 21, 1998. ADAMS Accession Number ML14300A038.

Hecla. 1992. Colorado Radioactive Materials License No. 317-02, Detection Monitoring Program for Groundwater Protection. Letter from G. Gamble (Hecla) to D. Simpson (CDH-RCD) May 29, 1992. Attachment in ADAMS Accession Number ML14055A291.

Hecla. 1991. Final Reclamation Plan Durita Site. Colorado Radioactive Materials License No. 317-02. Volumes 1 and 2. ADAMS Accession Numbers ML14300A089 and ML14300A095.

Hogan, J.F., F.M. Phillips, B.R. Scanlon, editors. 2013. Groundwater Recharge in a Desert Environment: The Southwestern United States. Water Sciences and Applications 9. American Geophysical Union, Washington District of Columbia.

IAEA. 2014. The Environmental Behavior of Radium: Revised Edition. Technical Reports Series No. 476. International Atomic Energy Agency, Vienna, Austria.

IAEA. 2010. Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Terrestrial and Freshwater Environments. Technical Reports Series No. 472. International Atomic Energy Agency, Vienna, Austria.

Landa, E.R. 2004. Uranium mill tailings: nuclear waste and natural laboratory for geochemical and radioecological investigations. Journal of Environmental Radioactivity 77, pp. 1-27.

McKinley, G., and A. Scholtis. 1993. "A Comparison of Radionuclide Sorption Databases Used in Recent Performance Assessments." Journal of Contaminant Hydrology, 13:347-363.

Meixner, T., A.H. Manning, D.A. Stonestrom, D.M. Allen, H. Ajami, K.W. Blasch, A.E. Brookfield, C.L. Castro, J.F. Clark, D.J. Gochis, A.L. Flint, K.L. Neff, R. Niraula, M. Rodell, B.R. Scanlon, K. Singham, M.A. Walvoord. 2016. Implications of projected climate change for groundwater recharge in the western United States. Journal of Hydrology 534:124-138.

Mitchell, N., D. Perez-Sanchez, and M.C. Thorne. 2013. A review of the behavior of U-238 series radionuclides in soils and plants. Journal Radiological Protection, 33:R17-R48.

Moo-Young, H., B. Johnson, A. Johnson, D. Carson, C. Lew, S. Liu, and K. Hancock. 2004. Characterization of infiltration rates from landfills: Supporting groundwater modeling efforts. Environmental Monitoring and Assessment 96:283-311.

NAS. 2007. Assessment of the Performance of Engineered Waste Containment Barriers. Committee to Assess the Performance of Engineered Barriers, Board on Earth Sciences and Resources, Division on Earth and Life Studies. National Academies Press, Washington, District of Columbia.

NRC. 2015. User's Guide for RESRAD-OFFSITE. NUREG/CR-7189, ANL/EVS/TM-14-2. U.S. Nuclear Regulatory Commission.

NRC. 2014. Letter from L. Dudes (NRC) to S. Tarlton (CDPHE) May 13, 2014 with Enclosure: Summary of the NRC's Review of the Draft CRR and Supporting Documentation. ADAMS Accession Number ML14092A206. U.S. Nuclear Regulatory Commission.

NRC. 2010. SA-900, Termination of Uranium Milling Licenses in Agreement States. ADAMS Accession Number ML101130527. U.S. Nuclear Regulatory Commission.

NRC, 2007. Assessment of Hydrogeologic Conditions and Need for Long-Term Groundwater Monitoring at the Former Durita Uranium Mill, Naturita, Colorado. ADAMS Accession Number ML071710354 or ML17243A316. U.S. Nuclear Regulatory Commission.

NRC. 2005a. Letter from J. Schleuter (NRC) to J. Vranka (CDPHE) December 2, 2005 with Enclosure: Durita Groundwater Review. ADAMS Accession Number ML17243A233. U.S. Nuclear Regulatory Commission.

NRC. 2005b. Letter from P. Lohaus (NRC) to S. Tarlton (CDPHE) February 10, 2005 with Enclosure: U.S. Nuclear Regulatory Commission (NRC) Review Comments of the Colorado Department of Public Health and the Environment's (CDPHE) Draft Completion Review Report (CRR) for the License Termination of the Durita Uranium Mill Site (RML 317-02). ADAMS Accession Number ML17243A2182. U.S. Nuclear Regulatory Commission.

NRC. 1996. RTM-96 Response Technical Manual, Volume 1, Revision 4. NUREG/BR-0150, U.S. Nuclear Regulatory Commission. ADAMS Accession Number ML091980341.

Prikryl, J.D. and D.A. Pickett. 2007. Recommended Site-Specific Sorption Coefficients for Reviewing Non-High-Level Waste Determinations at the Savannah River Site and Idaho National Laboratory. CNWRA Report. Center for Nuclear Waste Regulatory Analyses: San Antonio, Texas. ADAMS Accession Number ML073510127.

Robinson, R. 2004. Uranium Mill Tailings Remediation Performed by the US DOE: An Overview. Southwest Research and Information Center, Albuquerque, New Mexico. https://www.scribd.com/document/73586611/DOE-Overview-Uranium-Tailing-Remediation Accessed August 30, 2017.

Scanlon, B.R., D.G. Levitt, R.C. Reedy, K.E. Keese, and M.J. Sully. Ecological controls on water-cycle response to climate variability in deserts. Proceedings of the National Academy of Sciences. PNAS 102(17):6033-6038.

Schroeder, P. R., Aziz, N. M., Lloyd, C. M. and Zappi, P. A. 1994. The Hydrologic Evaluation of Landfill Performance (HELP) Model: User's Guide for Version 3. EPA/600/R-94/168a, U.S. Environmental Protection Agency Office of Research and Development, Washington, District of Columbia.

Serne, R.J. 2007. K_d Values for Agricultural and Surface Soils for Use in Hanford Site Farm, Residential, and River Shoreline Scenarios. Technical Report for Groundwater Protection Project – Characterization of Systems Task. PNNL-16531. Pacific Northwest National Laboratory, Richland, Washington.

Sheppard, M. I., and D. H. Thibault. 1990. Default Soil Solid/Liquid Partition Coefficients, K_ds, for Four Major Soil Types: A Compendium. Health Physics, 59(4)471-482.

Stonestrom, D.A., Constantz, J., Ferre, T.P.A., and S.A. Leake. 2007 Ground-Water Recharge in the Arid and Semiarid Southwestern United States. U.S.G.S. Professional Paper 1703. U.S. Geological Survey, Reston, Virginia. http://pubs.usgs.gov/pp/pp1703/.

Studious Solutions, 1994, Background Groundwater Quality and Approach for Comparison of Upgradient to Downgradient Concentrations at the Durita Site, Draft Proposal. March 29.

Vandenhove, H., C. Gil-Garcia, A. Rigol, and M. Vidal. 2009. New Best Estimates for Radionuclide Solid–Liquid Distribution Coefficients in Soils, Part 2: Naturally Occurring Radionuclides. Journal of Environmental Radioactivity 100:697–703.

Um, W., J.P. Icenhower, C.F. Brown, R.J. Serne, Z. Wang, C.J. Dodge, and A.J. Francis. 2010. Characterization of uranium-contaminated sediments from beneath a nuclear waste storage tank from Hanford, Washington: Implications for contaminant transport and fate. Geochimica et Cosmochimica Acta 74:1363–1380.

WRCC. 2017. Western Region Climate Center. Uravan station, Colorado. http://www.wrcc.dri.edu/summary/Climsmco.html. Accessed June 7, 2017.

Wright, M.T., K.G. Stollenwerk, and K.Belitz. 2014. Assessing the solubility controls on vanadium in groundwater, northeastern San Joaquin Valley, CA. Applied Geochemistry 48:41-52.

Yu, C., S.Kamboj, C. Wang, J.-J. Cheng. 2015. Data Collection Handbook to Support Modeling Impacts of Radioactive Material. ANL/EVS/TM-14/4. Argonne National Laboratory, Argonne, Illinois.

Zhao, M.-Y., Y.-F. Zheng, and Y.-Y. Zhao. 2016. Seeking a geochemical identifier for authigenic carbonate. Nature Communications 7:10885.

Zhu, C. 2003. A case against K_d-based transport models: natural attenuation at a mill tailings site. Computers & Geosciences 29(3), pages 351-359.