

Secondary Waste Form Development and Data Package for Hanford's Integrated Disposal Facility Performance Assessment – 16479

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ABSTRACT

The Hanford Low-Activity Waste (LAW) Vitrification facility is planned to immobilize Hanford tank waste for on-site, near-surface disposal at Hanford's Integrated Disposal Facility (IDF). During this process, liquid and solid secondary wastes will be generated which are also slated for disposal at the IDF. In order to receive authorization to dispose of the waste in the IDF, a Performance Assessment (PA) must be completed in accordance with DOE Order 435.1, Radioactive Waste Management, and its accompanying Manual. The PA must include calculations of potential doses to a member of the public for a 1,000-year period to provide a reasonable expectation that performance objectives will not be exceeded and that the waste will be managed in a manner that is protective of worker and public health and safety, and the environment. Liquid and solid secondary wastes are planned to be immobilized within cementitious waste forms similar to those used elsewhere in the United States and around the world for permanent disposal of low-level radioactive waste (LLW). Data packages and supporting documents are being developed to provide the technical underpinning for the analyses of contaminant releases to the environment from these waste forms for the IDF PA. Traditionally, releases from cementitious waste forms have utilized a diffusion-limited release model using contaminant-specific diffusivities measured in standard leach tests as inputs. More mechanistic modeling techniques are under development to increase the rigor of the PA analyses. In addition, recent testing has shown diffusivities of key contaminants can be reduced by several orders-of-magnitude through optimized formulation of the waste form and use of additives to bind specific contaminants.

INTRODUCTION

Radioactive byproduct wastes from nuclear weapons production are stored in underground tanks at the Hanford Site, located in the state of Washington. The waste tanks contain a complex and diverse mix of radioactive and chemical waste in the form of sludge, salts, and liquids, necessitating a variety of unique waste retrieval, treatment, and disposition methods. In general, the tank waste can be characterized as the following:

1. Sludge – Insoluble materials largely consisting of metal hydroxides and oxides that precipitated when acidic wastes from spent nuclear fuel

processing and other activities were neutralized and converted to high pH for storage in carbon steel tanks. The sludge fraction of the waste makes up the bulk of the material that will be processed via high-level waste (HLW) Vitrification into a stable glass form.

2. Supernatant – Liquid waste with high sodium content and high pH.
3. Saltcake – a mixture of salts that precipitated from supernatant as the concentration was increased by evaporation to reduce tank storage space requirements. Saltcake must be re-dissolved and processed as supernatant waste. The supernatant and saltcake contain the majority of highly radioactive cesium which must be separated and processed with the sludge stream into HLW glass. The decontaminated supernatant will then be processed via low-activity waste (LAW) vitrification into a stable glass form.
4. Potential contact-handled transuranic waste (CH-TRU): There are approximately 1.4 million gallons of waste in 11 specific single-shell tanks (SSTs). The material in these tanks is being reviewed to determine the potential for transfer to the Waste Isolation Pilot Plant (WIPP) versus being processed on-site into HLW and LAW glass fractions.

In order to begin immobilization of tank waste as soon as practicable, a Direct Feed LAW (DFLAW) flowsheet has been initiated. In the DFLAW configuration, LAW feed will be provided to the LAW Pretreatment System (LAWPS). The LAWPS will separate the HLW and LAW fractions and provide qualified feed to the Waste Treatment and Immobilization Plant (WTP) LAW Vitrification Facility. Liquid effluents from the WTP-LAW Facility will be managed in the DFLAW configuration by the WTP Effluent Management Facility (EMF).

Successful startup and operation of DFLAW requires the completion of engineering design and construction of numerous facilities, flowsheet stewardship, program integration across facilities, generation of a series of permits, and development of the regulatory framework to dispose of the waste forms generated. This paper discusses the need for data on non-glass waste forms to support a performance assessment (PA) of Hanford's Integrated Disposal Facility (IDF) to allow immobilized liquid and solid secondary wastes from DFLAW operations to be disposed there.

BACKGROUND

The Hanford LAW Vitrification facility is planned to immobilize waste feed for on-site, near-surface disposal at the IDF. During this process, liquid and solid secondary wastes will be generated which are also slated for disposal in the IDF, a RCRA-permitted mixed low-level waste (MLLW) disposal facility. In order to receive authorization to dispose of the waste, a PA must be completed in accordance with DOE Order 435.1, Radioactive Waste Management, and its accompanying Manual. The PA must include calculations of potential releases and doses to a member of

the public for a 1,000-year period after closure to provide a reasonable expectation that performance objectives will not be exceeded and that the waste will be managed in a manner that is protective of worker and public health and safety, and the environment.

The IDF will be used to dispose of both primary waste (vitrified LAW) and immobilized liquid and solid secondary wastes from waste treatment operations. Because the LAW glass is a highly durable waste form, release of contaminants is extremely slow and thus the secondary wastes are expected to represent a greater risk to human health and the environment. To support development of the PA, data on waste form performance are needed to model the long-term release of contaminants over the compliance period (1,000 years) and beyond (10,000 years or longer) to verify that impacts to groundwater will be within performance objectives, e.g. Environmental Protection Agency (EPA) drinking water standards at a well 100 meters down-gradient from the facility boundary.

To assess the long-term safety of an underground repository where the variability in processes, properties, and conditions is evolving and not completely understood, a hierarchical modeling approach has been adopted in the U.S., Europe, and other countries. At the top of the hierarchical modeling pyramid (Figure 1) is the total system PA (TSPA) model for decision-making, which must demonstrate that

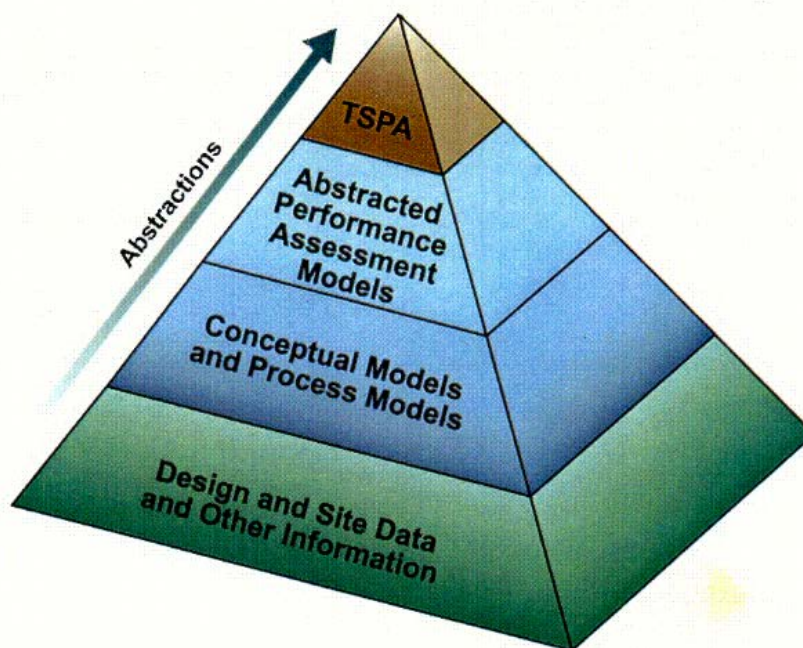


Figure 1. Total System Performance Assessment Information Pyramid (TRW 2000)

performance is acceptable when the uncertainties affecting performance have been accounted for. Monte Carlo methods used for the stochastic uncertainty analysis

are not tractable without simplifications to the representation and resolution of the modeled processes. The simplified modeling abstraction of the TSPA is intended to be conservatively representative but not necessarily mechanistic.

In the hierarchical modeling approach, the simplifications and assumptions in the TSPA model are based on mechanistically detailed field-scale and refined smaller scale models of flow, transport, and reactions. These modeling analyses are used to link relatively short-term monitoring and/or experimental observations with characterization data and databases in the development of technically defensible assumptions for long-term performance.

Multicomponent Reactive Transport Modeling

Within the last 15 years, the ability to integrate multicomponent reactions and rates into subsurface flow and transport simulators has enabled more comprehensive treatment of the geochemical barriers and attenuation processes afforded by cementitious waste forms and concrete structures surrounding waste forms. A critical use of these models is to address the time-dependent variation in the “constant” properties of the lumped parameter models (e.g., diffusion and distribution coefficients) used in the simplified TSPA model. In this case, more mechanistically detailed models may be used to identify how the lumped parameters vary as the hydro-geochemical conditions evolve. For example, release of constituents of concern (COCs) incorporated into waste form minerals may be modeled as a dissolution process subject to changing redox potential and/or pH. The use of a lumped parameter diffusion or equilibrium distribution coefficient (K_d) model does not account for the evolving geochemical conditions controlling COC release. It may be possible, however, to use a sequence of time periods, each with a different but constant diffusion coefficient and/or K_d , to represent the dissolution behavior. A multicomponent reactive transport simulator could be used to mechanistically address the evolving geochemical conditions, including pH and Eh, and to estimate representative lumped parameters that vary in time and space. Similarly, mineral precipitation and dissolution reactions that alter porosity, tortuosity, and permeability could be used to update the modeling of flow and transport processes.

Fracturing/Cracking of Cement Paste

A key difficulty in the prediction of long-term COC release from cementitious waste forms is accounting for fracturing and cracking. There are many features or processes that can lead to crack formation in cement, including chemical composition of the waste, the dry ingredients, hydration reactions and conditions, late stage ettringite formation due to reactions with sulfate, dissolution of $\text{Ca}(\text{OH})_2$ and calcium silicate hydrate (CSH) phases due to reactions with carbonate, and precipitation of calcite. While there are 2000-year old Roman concrete analogues (van der Sloot et al., 2000), there is general difficulty predicting rates for cement weathering, cracking, and turning to rubble over a 10,000-year time horizon. These

processes have important ramifications for the flow, transport, and reaction controls on COC release. Some complications typically associated with reactions in concrete (e.g., chloride corroding rebar, sulfate attack, alkali reactions with aggregate) will not likely be an issue with cement paste waste forms.

Previous Performance Assessments

The study of solidification and stabilization of Hanford waste using cementitious waste forms began in the 1980s, led by the Hanford Grout Disposal Program. The laboratory testing and field demonstration supported a PA that was produced in 1995 (Kincaid et al. 1995). However, the program was discontinued in the mid-1990s when vitrification was selected as the preferred alternative for treatment of Hanford LAW. Subsequent PAs (Mann et al., 1996, 2001, 2003a) evaluated glass as the LAW waste form initially to be disposed of in existing concrete vaults, then later in a lined burial trench which evolved to become the IDF.

In 2003, a risk assessment (RA) for supplemental LAW waste forms was prepared (Mann et al. 2003b). The purpose of the RA was to evaluate the long-term performance of ILAW glass and supplemental waste forms that were under consideration at that time (bulk vitrification, Cast Stone, and fluidized bed steam reforming) and the impacts of the release of contaminants to groundwater below the IDF. Because of budget, schedule, and technical limitations, Mann et al. (2003b) acknowledged that the RA was less rigorous and detailed than a PA, but had sufficient technical credibility to support decision-making.

The Tank Closure and Waste Management Environmental Impact Statement for the Hanford Site, Richland, Washington (TC&WM EIS; DOE/EIS-0391, 2012) evaluated a number of alternatives, including waste treatment options wherein liquid secondary waste, solid secondary waste, and supplemental LAW would be immobilized in a cementitious grout waste form. A diffusion-limited release model was used in the TC&WM EIS impact analyses to estimate the release of contaminants from the grouted waste forms. The diffusion-limited release model used contaminant-specific diffusivities and retardation factors along with waste form properties such as its bulk density (ρ), tortuosity (τ), constrictivity (δ), and effective porosity (ϵ) to calculate contaminant releases over a 10,000-year time period.

Although several analyses have been conducted over the last decade a formally reviewed PA has not been issued since the 2001 ILAW PA. Since that PA was completed the mission for the IDF has been expanded from accepting only ILAW glass to include other waste forms from WTP operations and other non-WTP waste forms. Also, construction of the IDF was completed in 2006 so data on the actual IDF location, design, and geology are available. The new PA, currently expected to be prepared in 2017, will utilize all of the available data and will provide the basis for IDF Waste Acceptance Criteria (WAC) and disposal authorizations needed to support DFLAW operations.

METHODS

Washington River Protection Solutions is conducting a waste form development and testing program to produce the data required for modeling the long-term performance of cementitious waste forms for the IDF PA. This program is a combined effort involving National Laboratories with expertise in waste form development. This work involves innovative formulations of waste forms tailored to the waste characteristics with additives selected to more tightly bind environmentally mobile constituents such as technetium-99 and iodine-129. Initial results from this work are promising and will be incorporated in a waste form performance data package for the IDF PA.

Wastes planned for disposal in the IDF will include LLW and MLLW from Tank Farm Operations and WTP Operations. The Interface Control Document calls for WTP to characterize solid secondary wastes from WTP operations, quantify inventories of COCs associated with those wastes, and recommend treatment for disposal. The IDF Waste Acceptance Criteria for a number of these solid secondary waste streams are expected to require grouting for mechanical stabilization and to provide encapsulation to minimize or retard the release of COCs. To the extent possible, data from the literature on cementitious barriers will be collected to allow PA modelers to predict long-term stability and release of contaminants from these waste forms.

Waste to be disposed of in the IDF will also include solidified wastes from treatment of liquid secondary wastes at Hanford's Effluent Treatment Facility (ETF). The ETF currently treats liquid effluents from various waste cleanup activities on the Hanford Site. In the current ETF flowsheet, contaminants in aqueous wastes are removed and/or destroyed through a series of filtration, ultraviolet oxidation, reverse osmosis, and ion exchange processes. The treated liquids are disposed in a state-approved land disposal site. Residual solids are dried in a thin-film dryer and packaged in 55-gallon drums for disposal.

Aqueous wastes to be treated in ETF are collected in one of three surface impoundments (Basin 42, Basin 43, and Basin 44) as part of the Liquid Effluent Retention Facility (LERF) in the northeast corner of the 200 East Area. Liquid wastes in the individual basins are processed through the ETF on a campaign basis. During DFLAW operations, the wastes to be processed through ETF will include WTP secondary liquid wastes, liquid wastes from the LAW pretreatment system, 242-A evaporator condensates, leachates from the IDF and mixed waste burial trenches, and other miscellaneous wastes (May et al. 2009).

Cooke and Lockrem (Cooke and Lockrem 2005; Cooke et. al. 2006), conducted a screening study of hydraulic binder formulations for solidification of LERF Basin 42 wastes composed primarily of 242-A evaporator condensates. In their initial work,

they used a dry blend of 10 wt% cement, 44 wt% fly ash¹, and 46 wt% blast furnace slag (BFS). A calcium aluminate (SECAR 51^{®2}) was added to bind sulfate in the wastes as calcium sulfate to accelerate the formation of ettringite, which, if the ettringite forms late in the curing process, can cause swelling of the waste form, leading to cracking. In the next phase of testing, Cooke et. al. (2006) tried a number of mixes with portland cement, BFS, fly ash, and hydrated lime as an alternate source of calcium to facilitate the rapid ettringite formation. Their final recommended formulation was for a dry blend mix of 36 wt% portland cement, 36 wt% BFS, and 28 wt% hydrated lime.

In FY15, a team lead by WRPS, PNNL, and SRNL planned a matrix of test formulations following the recommendations of Cooke et. al. (2005). Tests were conducted on treated ETF waste simulants, including DFLAW Vitrification off-gas condensate, 242-A evaporator condensates, and LLW disposal trench leachates. Testing included cementitious waste forms using a blend of portland cement, BFS, and hydrated lime with the standard Cast Stone dry blend mix as a control. Cast Stone was developed at Hanford in early 2000s and consists of 45 wt% fly ash, 47 wt% BFS and 8 wt% portland cement. Waste form characteristics included properties, such as hydraulic conductivity, porosity, density, and water retention curves (i.e. matric potential) for use in contaminant release modeling as well as effective diffusivity measurements via EPA Method 1315, Tc solubility, and Tc desorption K_ds that may also be used for modeling near-field contaminant releases from the waste forms for PA analyses.

A rigorous framework for modeling the long-term performance of a waste form utilizes a mechanism in which matrix hydrolysis and contaminant release are controlled by the rate that chemical bonds are broken. Such a mechanistic approach has been successfully employed for vitrified waste forms (McGrail et al. 2000). While similar arguments can be made regarding the importance of modeling chemical reactions and transport in cement pore waters (Bacon et al. 2002), with cementitious waste forms, a physical model of contaminant diffusion has been almost universally adopted (Cook et al. 2005; Serne and Westsik 2011).

Diffusional release of species from cementitious waste forms is best treated as a combination of physical transport and chemical interactions. The fundamental basis for the diffusion model is Fick's Second Law wherein mass transport into or out of a porous media is expressed in terms of component-specific diffusion coefficients. For each species, the apparent diffusion coefficient for a porous media (D_a) is a measure of the physical contribution to diffusion, and depends on the molecular diffusion coefficient (D_o) of a particular solute in dilute solution, and the tortuosity (τ) constrictivity (δ) and porosity (ϵ) of the porous medium:

$$D_a = D_o \frac{\epsilon \delta}{\tau^2} \quad (1)$$

¹ Although not specified in the report, it was assumed that the fly ash used was Class F as defined in ASTM C 618

² SECAR is a registered trademark of Lafarge Calcium Aluminates, Inc., Chesapeake, VA.

Chemical interactions can be quite varied (adsorption, ion exchange, precipitation, specific and irreversible adsorption), and each process may have fast or slow kinetics. The simplest process that is mathematically tractable is reversible adsorption with fast kinetics and a linear isotherm. This simple chemical process can be described using the equilibrium distribution coefficient K_d :

$$K_d = \frac{C_s}{C} \quad (2)$$

where C_s is the concentration of the solute of interest in the solid and C is the concentration in solution. For this relationship to be accurate, the chemical reaction processes it is intended to describe must be fast and reversible and the sorption process satisfies the linear isotherm constraint. Chemical reactions for contaminants do not always meet these requirements. Regardless, this simple construct is often applied in quantifying the release of contaminants from cementitious waste forms because it allows one to separate the physical and chemical processes that control transport of contaminants. The impact of the chemical interactions of the solute with the porous media on the effective diffusion coefficient is described by the following equation:

$$D_e = \frac{D_a}{R} \quad (3)$$

where D_a is the apparent diffusion coefficient and R is the retardation factor. The retardation factor is related to the K_d by the following equation:

$$R = 1 + \frac{\rho_b K_d}{\varepsilon} \quad (4)$$

where ρ_b is the bulk density of the porous solid waste form and ε is its porosity. There are several experimental methods that one can use to measure the K_d and then compute R after measuring the porosity and bulk density of the waste form. Conversely, one can measure the apparent diffusion coefficient using non-reactive species and through diffusion cells, penetration profiles of a contaminant into a solid porous medium, or out diffusion of contaminants (leaching tests).

Empirical diffusion coefficients measured in short-term laboratory experiments have been widely used in modeling long-term performance of cementitious waste forms (Albenesius 2001). These procedures have changed little since the International Atomic Energy Agency (IAEA) method was proposed by Hespe (1971) over 40 years ago. Effective or observed diffusion coefficients for each contaminant have been used for diffusion-controlled transport analysis of both intact cementitious monolith waste forms emplaced in subsurface porous environments and fractured cementitious waste forms where contaminant release is dominated by diffusive-advective processes.

Because only limited waste-form-specific data for secondary wastes solidified in cementitious grouts are available at this time, it may be prudent to utilize the simple diffusion-based modeling approach for the IDF PA. It should be noted,

however, that other mechanisms of cementitious waste form degradation and contaminant release are being considered and incorporated in the Savannah River Site saltstone PA; for example, solubility controlled release of technetium under reducing conditions, sulfate attack, carbonation, and redox capacity reduction through oxygen influx (SRR CWDA 2014).

RESULTS AND DISCUSSION

Previous analyses have shown the primary constituents of potential concern (COPCs) in solid and liquid secondary wastes to be disposed of in the IDF include Tc-99, I-129, U, Cr, NO_3^- , and NO_2^- . The primary purpose of the PA analyses is to model the fate and transport of the radionuclides in order to estimate future radiation doses to members of the public. However, the same models may be used to predict chemical exposures and concentrations relative to health-based limits for RCRA permitting purposes.

Much of the data needed for PA modeling will be derived from selected tests from the Leaching Environmental Assessment Framework (LEAF) which is a collection of four leaching tests that can be used to develop a characteristic leaching profile of the subject material under equilibrium- and mass transfer-controlled release conditions (Kosson et al., 2002). Testing for the Hanford Liquid Secondary Waste Treatment program has focused primarily on EPA Methods 1313 and 1315 supplemented with Tc adsorption/desorption K_d and Tc empirical solubility measurements on the grout waste form.

EPA Method 1315 is a tank-type monolith immersion leach test similar to predecessor methods ANSI/ANS 16.1 and ASTM C1308 and is most useful for measuring observed (effective) diffusivities of COPCs such as Tc-99, I-129, and Cr. Previous work by the Pacific Northwest National Laboratory (PNNL) (Mattigod, et al. 2011) using cementitious and hybrid geo-polymer waste forms made from simulated liquid secondary wastes demonstrated that each of these methods yielded essentially equivalent results allowing cross-comparison of results from current testing programs to results from prior work extending back years or even decades.

Based on the 2015 data package for cementitious waste forms, values of effective diffusion coefficients suitable for use in PA analyses are presented in Table 1. Recent data taken from the ongoing testing is included in the table with 'to be determined' indicating that an update will be released in 2016 for use in the upcoming IDF PA.

Table 1. Effective Diffusion Coefficients for Cementitious Waste Forms

Contaminant	Effective Diffusion Coefficient (cm ² /s)		
	Previously Recommended ^a		Recent data
	Secondary Waste ^a	LAW Waste ^a	Secondary Waste
Technetium	2.0 × 10 ⁻¹⁰ – 6.0 × 10 ⁻¹²	2.0 × 10 ⁻¹⁰ – 6.0 × 10 ⁻¹²	3.2 × 10 ⁻¹³ – 1.2 × 10 ⁻¹⁴
Iodine	1.0 × 10 ⁻⁸ – 2.0 × 10 ⁻⁹	5.0 × 10 ⁻⁸ – 1.0 × 10 ⁻⁸	Not measured
Sodium	1.0 × 10 ⁻⁸ – 2.0 × 10 ⁻⁹	2.5 × 10 ⁻⁸ – 7.9 × 10 ⁻⁹	To be determined
Nitrate	1.0 × 10 ⁻⁸ – 2.0 × 10 ⁻⁹	4.0 × 10 ⁻⁸ – 7.9 × 10 ⁻⁹	To be determined
Nitrite	1.0 × 10 ⁻⁸ – 2.0 × 10 ⁻⁹	4.0 × 10 ⁻⁸ – 6.3 × 10 ⁻⁹	To be determined
Chromium	9.0 × 10 ⁻¹³ – 8.0 × 10 ⁻¹⁵	9.0 × 10 ⁻¹³ – 8.0 × 10 ⁻¹⁵	Not measured
Uranium	<6.0 × 10 ⁻¹⁶	<6.0 × 10 ⁻¹⁶	Not measured

^a – from Cantrell 2015

Recent testing has shown diffusivities of some key contaminants can be reduced by several orders-of-magnitude through optimized formulation of the waste form and use of additives to bind the contaminants. Specifically effective diffusivities for Tc-99 ranging from 10⁻¹³ to 10⁻¹⁴ cm²/s were measured in recent tests of cementitious waste forms made with Hanford Liquid Secondary Waste simulants. This compares favorably to diffusivities of 10⁻¹¹ cm²/s achieved without formulation enhancements and conservative values used in previous analyses of 5 × 10⁻⁹ cm²/s. In recent physical properties testing hydraulic conductivity of 3 × 10⁻⁹ cm/s was measured indicating low permeability similar to that observed for other cementitious waste forms such as Savannah River Site saltstone.

Although diffusion is the primary mechanism used to describe contaminant release from cementitious waste forms, it is likely that under certain circumstances, solubility constraints will control release of specific contaminants, such as Tc, Cr, and U. This would be particularly true under low-flow conditions and low redox (reducing) environments. The effect of reduced solubility is manifested in leaching experiments as a reduced effective diffusion coefficient. In order to discreetly model this effect, the solubility of the contaminant in the waste form must be measured. Recently, it has been demonstrated that under reducing conditions, Tc release from Saltstone is controlled by TcO₂·xH₂O solubility (Cantrell and Williams 2013).

A further refinement of contaminant transport modeling from cementitious waste forms involves modeling the retardation of contaminant release due to adsorption/desorption between the contaminant and the waste form solids. This requires knowledge of contaminant distribution coefficients (K_ds) in contact with the waste form material. Actual K_d data for key contaminants specific to Hanford's immobilized liquid secondary waste were not available and thus data from SRR CWDA 2014 were provided as interim values (see Table 2). Testing is ongoing to

obtain K_d s for Tc in Hanford cementitious waste forms. The data will be provided in an update to the cementitious waste forms data package.

Table 2. Desorption K_d (mL/g) Values for Cementitious Materials
(from SRR CWDA 2014 Table 4.1-4)

Component	Reduced Region	Oxidized Region II	Oxidized Region III	Reference
Tc	Solubility Control	0.5	0.5	a
I	9	15	4	a
Na	1	1	0.5	a
Cr	1000	10	1	a
U	2500	1000	100	b

^aKaplan (2010), ^bSeaman and Kaplan (2010)

Finally, in order to model flow and transport of contaminants from the waste form, fundamental physical properties are needed that largely govern the rate at which the vadose zone pore water percolating through the disposal facility can flow into and through the waste form. Example values appear in Table 3 below. The first five in the list are parameters determined or used in previous PAs for saltstone and the Hanford Grout Program. The last two in the list are candidate waste forms for disposal in the IDF and data will be updated when available.

Table 3. Physical Properties of Cementitious Waste Materials

Waste Type	Porosity (%)	Dry Bulk Density (g/cm ³)	Particle Density (g/cm ³)	Hydraulic Conductivity K_{sat} (cm/s)	Source
DDA (deliquification, dissolution and adjustment)	0.55	1.06	2.37	9.6×10^{-11}	a
ARP/MCU (Actinide Removal Process/Modular Caustic Side Solvent Extraction Unit)	0.59	0.97	2.38	8.5×10^{-10}	a
SWPF (Salt Waste Processing Facility)	0.58	1.01	2.42	6.0×10^{-09}	a
All wastes in saltstone PA	0.58	1.01	2.40	6.4×10^{-09}	b
Hanford DSSF (double-shell slurry feed)	-	1.10	2.61	2.3×10^{-08}	c
Immobilized liquid secondary waste	0.53-0.56	1.2-1.3	2.7	3×10^{-09}	d
Encapsulation barrier grout	TBD	TBD	TBD	TBD	

^aSerne and Westsik (2011), ^bSRR CWDA (2014), ^cRockhold et al. (1993), ^d – Cozzi et al. (2015)

These values are for freshly cured laboratory samples. The PA analyses will need to consider how these properties may change over long periods of time, e.g. hundreds

to thousands of years. Methods to address the evolution in grout properties at long weathering times range from a simplistic assumption that the waste form degrades to rubble instantly after 500 years to mechanistic modeling of changes in waste form properties over time due to oxidation, carbonation, sulfate attack, etc.

CONCLUSIONS

The Hanford LAW Vitrification facility is planned to immobilize waste feed for on-site, near-surface disposal at Hanford's IDF. During this process, liquid and solid secondary wastes will be generated, which are also slated for disposal in the IDF. Liquid and solid secondary wastes are planned to be immobilized within cementitious waste forms similar to those used elsewhere in the United States and around the world for permanent disposal of LLW. Data packages and supporting documents are being developed to provide the technical underpinning for the analyses of contaminant releases to the environment from these waste forms for the IDF PA. Traditionally, releases from cementitious waste forms have utilized a diffusion limited release model using contaminant-specific effective diffusivities measured in standard leach tests as inputs. More mechanistic modeling techniques are under development to increase the rigor of the PA analyses.

Results of ongoing testing of Hanford cementitious waste forms, which will be in the updated data package for the upcoming IDF PA, indicate improvement (as indicated by the diffusion coefficients listed in Table 1) by using improved dry blend formulations and additives to bind contaminants. Work on the enhancements of cementitious waste forms will continue. Also, as more mechanistic modeling techniques are pursued, more data will be needed on different waste form characteristics and behaviors. Hence the cementitious waste form testing program is expected to continue for the foreseeable future to accommodate such data needs.

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