

MODEL ABSTRACTION OF WASTE FORM DEGRADATION IN ALTERNATIVE DISPOSAL SITE

T. Ahn¹, H. Jung², E.L. Tipton², and T. Sippel¹

¹U.S. Nuclear Regulatory Commission (NRC), Division of High-Level Waste Repository Safety, Washington, DC 20555, USA

²Center for Nuclear Waste Regulatory Analyses (CNWRA), 6220 Culebra Road, San Antonio, Texas 78238-5166, USA

Contact: Tae Ahn, tae.ahn@nrc.gov

The U. S. Nuclear Regulatory Commission and its contractor the Center for Nuclear Waste Regulatory Analyses staff (CNWRA) have been developing a beta (preliminary) version of the performance assessment model Scoping of Options and Analyzing Risk (β -SOAR) model. This model is intended to provide risk and performance insights for a variety of potential geological disposal concepts for high-level radioactive waste and spent nuclear fuel. The model considers the radionuclide mobilization processes from waste form dissolution that affect the source term (i.e., the amount and type of radionuclides released from the waste form for radionuclide transport). The waste forms are emplaced inside a conceptual waste package and may be covered by cladding materials. This paper presents the Waste Form Model Component (WFC) in the β -SOAR model, that specifically considers the type of solid waste form, important radionuclides, initial radionuclide inventory, and degradation of the waste form following exposure to the aqueous environment after the waste package (WP) breaches. The WFC receives WP failure times and breach fraction of WP from the Waste Package Component. The WFC provides the amount of radionuclides mobilized to the Near Field Component NFC.

I. INTRODUCTION

Recently, the U.S. Nuclear Regulatory Commission (NRC) has been developing a computer model, *Beta-Scoping of Options and Analyzing Risk (β -SOAR)*. The β -SOAR model is designed to provide the NRC staff timely risk and performance insights for a variety of potential high-level radioactive waste (HLW) spent nuclear fuel (SNF) disposal options (NRC and CNWRA, 2010). The source term of the β -SOAR is developed primarily based on the type of waste form, important radionuclides, initial radionuclide inventory, and degradation rate of various waste forms. This paper describes how relatively complex process models related to source term are abstracted for the simplified model for system implementation. The abstracted model was exercised with the β -SOAR to check its reasonableness with respect to the characteristics of potential waste forms and disposal environments.

The Waste Form Component (WFC) model of β -SOAR considers the radionuclide mobilization processes from waste form dissolution, which affect the source term for radionuclide transport. The WFC specifically considers the type of solid waste form, important radionuclides, radionuclide inventory, and degradation of the waste form following exposure to the aqueous environment after waste package breaches. The WFC receives waste package failure times from the Waste Package Component and breach fraction, and provides the amount of radionuclides mobilized to the Near Field Component.

II. WASTE FORM COMPONENT

II.A Model Implementation

The WFC model computes the amount of radionuclides dissolved from the waste form and mobilized to the NFC in aqueous environments. In the present model, the β -SOAR considers two distinct waste forms: SNF and HLW glass associated with reprocessing the SNF. The SNF separates commercial SNF from spent mixed-oxide fuel (sMOX). An initial inventory contains 15 radionuclides (i.e., Pu-238, 239, 240, 242, U-232, 233, 234, 235, 236, 238, Np-237, C-14, Cs-135, I-129, Tc-99) bound on the SNF (or glass) matrix. Each bound radionuclide inventory has its own dissolution rate. These radionuclides were chosen to represent a variety of geochemical behaviors (e.g., mobilization) and potential waste form inventories for commercial SNF, HLW glass, and sMOX waste forms, based on the performance assessment results of various waste forms in both oxidizing and reducing environments (Bechtel SAIC Company, LLC, 2004; Leslie et al., 2007; Marivoet, et al., 2001). In addition to the bound radionuclide inventories, the SNF radionuclides are further segregated into unbound inventories for C-14, Cs-135, I-129, and Tc-99. These unbound inventories are rapidly released from the SNF waste forms upon contact with water and are used to more conservatively model release of these radionuclides through other mechanisms (i.e. gap and grain inventories). Each waste package contains a mixed (blended) waste form where the user may specify the ratio of commercial SNF, HLW glass and sMOX.

For the bound radionuclide inventories, the total amount of mobilized radionuclides (M) from waste form dissolution equals the fractional mobilization rate (m) multiplied by total inventory of radionuclide (I) and the time of waste form contacting aqueous environment (t)

$$M [\text{g}] = m [\text{fraction year}^{-1}] \times I [\text{g}] \times t [\text{year}] \quad (1)$$

where, the fractional mobilization rate (m) equals dissolution rate (R) multiplied by specific surface area (A).

$$m [\text{fraction year}^{-1}] = R [\text{g cm}^{-2} \text{ year}^{-1}] \times A [\text{cm}^2 \text{ g}^{-1}] \quad (2)$$

II.B. Key Model Assumptions

- Fission products are assumed to be released at the fractional mobilization rate, and actinides are assumed to be released at their solubility limits (Wilson and Gray, 1990). The segregated radionuclides are conservatively assumed to be released instantaneously after the waste form contacts water.
- The dissolution rate of sMOX is assumed to be represented by commercial SNF, based on limited literature data on sMOX. The radionuclide inventory of sMOX is separated from commercial SNF.
- The dissolution process is redox-sensitive for commercial SNF and sMOX due to the nature of the corrosion process. Dissolution of HLW glass is not redox-sensitive because dissolution (leaching) of HLW glass will be a chemically controlled process. Although HLW glass dissolution is mainly dependent on pH and temperature, these dependencies are accounted for by the range of input values associated with dissolution rates.
- By the time waste package breaches occur in the potentially reducing repository, it is assumed that an initial disturbed oxidizing condition would be replaced by a reducing environment, with residual radiolysis products (e.g., hydrogen peroxide, an oxidizer) present from long-lived actinides.

II.C. Default Parameters

The default parameters in the WFC include radionuclide inventory, reprocessing effect on inventory, instant release fraction of fission and activation products, and fractional rate of waste form mobilization.

The data used in the model are primarily based on the published data on waste forms from domestic and international repository programs. The independent NRC

data available are extensively used as the initial input values for several parameters.

- **Radionuclide Inventory:** initial radionuclide inventories consider that a waste emplacement by geologic disposal occurs 50 – 100 years before the disposal site closure to consider reversibility (or retrievability). Important radionuclide inventories are estimated from characterization of commercial SNF, sMOX, and HLW glass. The 15 radionuclides used in the WFC model were selected in order to be representative of radionuclide transport in both oxidizing (e.g., alluvium) and reducing environments (e.g., bentonite, clay and tuff). In an oxidizing environment, the contribution to dose from actinides is expected to be significant due to relatively high solubility limits in the near field; however, contribution to the dose from the unbound fission products are expected to be significant. In a reducing environment, the contribution to dose from actinides is expected to be low with a relatively low solubility limits in the near field (as compared to those in an oxidizing environment). Mobilization of fission and activation products are redox insensitive in HLW glass and redox sensitive in commercial SNF and sMOX. Therefore, fission and activation products will likely to be more mobile in the reducing environment. This is especially important for reprocessed waste forms because fission and activation products would likely have lower extraction potential. Inventories are defined on a per waste package basis. The number of waste packages is 10,000, based on waste loading of current commercial SNF or HLW glass. The amounts of radionuclides are referenced from the domestic data base related to high-level nuclear waste disposal program.
- **Reprocessing Effect on Inventory:** the WFC model uses a mixed (blended) waste form inventory which is determined by combining proportions of the commercial SNF, sMOX, and HLW glass radionuclide inventories. This capability allows the user to readily explore various combinations of waste form inventories (including those that could arise from reprocessing). The user may select the waste form inventory by specifying values for two parameters: the fraction of the waste inventory that is SNF (the fraction composed of commercial SNF and spent mixed-oxide fuel, the remaining fraction is assumed to be HLW glass) and the ratio of SNF that is commercial SNF (the remainder is assumed to be sMOX). For example, if the user sets the value of the fraction of SNF to be 0, the representative waste form would be entirely the HLW glass radionuclide inventory. Likewise, the user could specify a waste

form as entirely commercial SNF or sMOX by setting the value of the fraction of SNF to 1, and the ratio of commercial SNF to sMOX to a value of 1 or 0, respectively.

- **Instant Release Fraction of Fission and Activation Products:** the distribution of radionuclides is also characterized, including homogeneous distribution in the matrix (the bound inventory) and relatively inhomogeneous distribution through segregation or compound formation (the unbound inventory; e.g. radionuclides present in the gap and grain boundaries of SNF). In the present WFC model, an instant release fraction of fission and activation products from the initial radionuclide inventory is set aside for immediate release upon failure of a waste package. This is done to more conservatively account for radionuclides that may be present in the gap and grain structures of both commercial SNF and sMOX, allowing for rapid release upon exposure of the fuel matrix to water. The instant release fractions of commercial SNF and sMOX used are the values in the NRC TPA 5.1 User Guide (Leslie et al., 2007; Jain et al., 2004). The values are given in a triangular distribution because the most expected value is not necessarily a mean value. For example, most of high burnup (> 45 GWd/MTU) commercial SNF may have more instant release fractions than matrix releases (Jain et al., 2004). Nevertheless, the high burnup commercial SNF would be of a smaller overall abundance compared with commercial SNF of lower burnup (< 45 GWd/MTU).
- **Mobilization of Waste Form:** In the commercial SNF, radionuclides will dissolve at the same rate of UO₂ matrix dissolution. In an oxidizing environment, UO₂ matrix will dissolve electrochemically in soluble species with an aid of oxidants such as dissolved oxygen and hydrogen peroxide (Shoesmith, 2000). In a reducing environment, UO₂ matrix will dissolve chemically in soluble species (Sunder and Shoesmith, 1991). The dissolution of sMOX is assumed to be represented by commercial SNF due to limited data and potential similar UO₂ matrix. In the HLW glass, radionuclides are assumed to dissolve at the dissolution rate of HLW glass matrix. The HLW glass matrix will dissolve chemically where the process is controlled by concentration of dissolved silica compound in both oxidizing and reducing environments (Bechtel SAIC Company, 2004). The process of radionuclide mobilization varies depending on the type of the radionuclide distribution. The segregated, unbound radionuclides are mobilized rapidly after water contacts the waste form. The bound radionuclides in the matrix or compounds are mobilized from the

dissolution of waste forms in the aqueous environment. The fractional mobilization rates are assumed to be constant within uncertainty ranges at a given temperature. The environmental conditions are important in determining specific dissolution rates, including near field water chemistry, temperature or redox (e.g., reducing or oxidizing) conditions. The dissolution rates of the waste form in the β -SOAR are referenced from multiple domestic and international literature data and are given depending on the redox condition (i.e., oxidizing or reducing) or as a combined value determined by normalizing those two redox conditions to consider the radiolysis effect. Additionally, other environmental effects (e.g., groundwater chemistry or pH) are accounted for by using a broad range of dissolution rates, based on the dominant environmental conditions in the international disposal systems. A log uniform distribution has been used to represent the fractional mobilization rate. The distribution accounts for uncertainties in fractional mobilization rates due to various environmental conditions and the range of values selected for the lower and upper ends of the distribution bound those reported in the international literature. In connecting the dissolution rate to the fractional mobilization rate (Equation 1-2 above), the specific surface area is determined by the average fragment size (radius) and density of waste form. In the present model, two different fragment sizes are used: 0.1 cm [0.04 inch] for commercial SNF (NRC, 2008) and 10 cm [3.9 inch] for HLW glass (Bechtel SAIC Company, 2004). Additionally, other factors such as surface roughness (NRC, 2008) were considered in the calculation of the fractional mobilization rate.

II.D. Model Flexibility

The β -SOAR model has flexibility to be run primarily for different types of waste form, different types of environments including redox conditions (e.g., oxygen or radiolysis effects), various waste loading, and more radionuclide types. A user defined waste form is available and can be modified for defining specific radionuclide inventories and fractional mobilization rates. Additionally, the user may modify the fractional rate of waste form mobilization rate by means of a multiplier. This parameter could be used to approximate mostly inhibiting effects on the waste form mobilization. For example, there could be protection by partially failed cladding, or redox variation or radionuclide sorption with iron or iron compounds. The user may also specify a fraction of the initial radionuclide inventory that is set aside and not available for mobilization (e.g., to simulate intact cladding). The effects of temperature on the fractional mobilization rate can also be modeled using an

Arrhenius relation for commercial SNF and HLW glass. The values for activation energy in the model are from the NRC TPA 5.1 User Guide (Leslie et al., 2007) for commercial SNF and domestic literature data for HLW glass. The temperature corrections using the activation energies are made in the abstraction of dissolution rates.

II.F. Reduction of Uncertainty

Processes in the WFC model are highly abstracted, in part to account for the broad range of options being considered for disposal. Nevertheless, future improvements to this abstraction could reduce uncertainties in several areas:

- Ceramic or metallic waste forms may occur. These waste forms have not been well characterized or studied. The waste loading and the number of waste packages may need to be optimized to limit heat generated. Increasing waste loading may result in more heat generation. Similarly, sMOX and higher-loading HLW glass do not have sufficient data and process-level models at this time, to be abstracted in the β -SOAR model.
- More different types of radionuclide could be considered. They may include Cl-36, Se-79, Zr-93, Pd-107, Sn-126, Ra-226, Ac-227, Pa-231, and Th-(229, 230).

- The properties of waste form could be better formulated. For example, the size of fragment (or chunk) of waste form is important in converting the dissolution rate to the fractional mobilization rate.
- The temperature and redox relationships as a function of time could be implemented for WFC and other engineered components such as various waste package types and backfill conditions.
- When the disposal environment is better defined, the model could be refined in terms of redox, temperature, and groundwater chemistry. This may be especially important for a potential disposal system hosted in salt. Salt rock has unique features in terms of aggressiveness of water chemistry, availability of aqueous condition, and higher temperatures.
- Partial protection by internal structure of the waste package could be more elaborated. For example, the amount of intact cladding could be quantified. The instant release could be consider the degradation of the internal structure containing the radionuclide (e.g., C-14).

III. INPUT PARAMETERS

Parameter Name	Value (g)	Description and Basis
Initial Inventory of Commercial SNF (gram per waste package, 1 g = 0.035 oz)	C14, 1.35E+00 Cs135, 4.36E+03 I129, 1.73E+03 Np237, 1.54E+04 Pu238, 1.52E+03 Pu239, 4.32E+04 Pu240, 2.05E+04 Pu242, 5.28E+03 Tc99, 7.55E+03 U232, 1.02E-02 U233, 5.76E-02 U234, 1.75E+03 U235, 6.26E+04 U236, 3.84E+04 U238, 7.82E+06	The initial radionuclide inventory for Commercial SNF was based on the inventory in DOE (2008) Table 2.3.7-3 as an example case. The inventory was modified to account for the eventual decay of Pu-241 and Am-241 (not tracked in the β -SOAR) into Np-237, by adding the initial values for Pu-241 and Am-241 to the initial value for Np-237 this adjustment prevents the effect of Np-237 from being underestimated.
Initial Inventory of HLW Glass (gram per canister, 1 g = 0.035 oz)	C14, 2.91E-01 Cs135, 9.47E+02 I129, 9.36E+01 Np237, 5.06E+02 Pu238, 5.31E+01 Pu239, 3.43E+02 Pu240, 3.86E+01 Pu242, 5.44E+00 Tc99, 1.36E+0	The representative HLW glass radionuclide inventory was based on the radionuclide inventory in DOE (2008) Table 1.5.1-21 as an example case. The radionuclide inventory was derived by selecting the highest value for each radionuclide out of the four glass waste forms (Hanford, SRS, West Valley, and Idaho) presented in the table. Additionally, the initial inventory of Pu-241 and Am-241 are added to the initial inventory of Np-237 to account for in-growth.

	U232, 1.47E-03 U233, 9.39E+00 U234, 1.79E+01 U235, 3.07E+02 U236, 5.69E+01 U238, 1.41E+05	
Initial Inventory of sMOX (gram per waste package, 1 g = 0.035 oz)	C14, 2.58E+00 Cs135, 7.17E+03 I129, 3.09E+03 Np237, 4.64E+04 Pu238, 1.50E+03 Pu239, 1.03E+05 Pu240, 7.26E+04 Pu242, 1.84E+04 Tc99, 9.98E+03 U232, 1.01E-03 U233, 2.33E-03 U234, 9.13E+01 U235, 5.62E+03 U236, 2.23E+03 U238, 8.19E+06	The initial radionuclide inventory for sMOX was based on the inventory in Table 6-2 (Sandia National Laboratories, 2007a) as an example case. The inventory was modified to account for the eventual decay of Pu-241 and Am-241 (not tracked in the β -SOAR) into Np-237, by adding the initial values for Pu-241 and Am-241 to the initial value for Np-237.
Mobilization (Degradation) of Commercial SNF and sMOX under the Oxidizing Condition (fraction per year; minimum and maximum)	3.00E-05, 6.00E-04 (log-uniform)	The oxidizing environment is considered because of the potential alpha radiolysis in the reducing environment and the early waste package failure. The assessment is more based on immersion conditions that are considered in the alternative disposal sites in the future (NRC, 2008). The dissolution rate of Commercial SNF is assumed to bound that of sMOX under immersion conditions (DOE, 2003). Both of commercial SNF and sMOX have the particle size of ~ 1 mm after reactor irradiation. Other references include Leslie et al. (2007) and Shoemith (2000).
Mobilization (Degradation) of Commercial SNF and sMOX under the Reducing Condition (fraction per year; minimum and maximum)	9.00E-07, 2.00E-05 (log-uniform)	An average factor of 0.03 (from 0.01 – 0.1) was factored in the oxidizing case. In the French and Belgian repositories, an average 2×10^{-6} /year was used (Iseghem, 2007), similar to the current estimate. To be consistent, the dissolution rate of sMOX was assumed to be same as the rate of commercial SNF (DOE, 2003).
Mobilization (Degradation) of Commercial SNF and sMOX under the Combined Condition (fraction per year; minimum and maximum)	9.00E-07, 6.00E-04 (log-uniform)	Because the alpha radiolysis may have limited effects on the dissolution rate of commercial SNF (Ferry, et al., 2005) and sMOX, the combined case is separated to represent some effects of alpha radiolysis. If we consider the hydrogen effects to be produced by the container corrosion, this combined rate could be conservative. The hydrogen could inhibit the SNF dissolution rate (Broczkowski, et al., 2005). To be consistent, the dissolution rate of sMOX was assumed to be same as the rate of commercial SNF (DOE, 2003).
Mobilization (Degradation) Rate of HLW Glass (fraction per year; minimum and maximum)	1.50E-06, 2.00E-04 (log-uniform)	French and Belgian HLW glass determined the fractional mobilization rate, 1.4×10^{-5} per year (Iseghem, 2007). Other appropriate data include fractional mobilization rate, 1.4×10^{-5} per year (DOE, 2009, SAR RAI Response Table 1.1-2) at pH = 7.5. Appropriate pH ranges are: longer-term pH ~ (8.0 ~ 8.5) (Rosborg and Wermer, 2008); and immersion pH ~ 8.0 (SNL, 2007b) as L/year increases. HLW glass has higher dissolution rates per unit area compared

		with SNF [(in equations (1) and (2)]. However, in the actual calculation of fractional mobilization per year, a bigger chunk of ~ 10 cm was used (Bechtel SAIC Company, 2004), compared with ~ 1 mm of commercial SNF or MOX SNF. This decreases the fractional release rate by a factor of ~100.
Unbound Radionuclide Fraction in Commercial SNF and sMOX	Cs-135, 0.0088 (lower), 0.025 (apex), 0.058 (upper) I-129, 0.0054, 0.11, 0.32 Tc-99, 0.0005, 0.0013, 0.0028 C-14, 0.1 (constant value)	All are measured values (Leslie et al., 2007; Jain et al., 2004). The values used for the radionuclide unbound fractions for commercial SNF were also used for sMOX due to limited data available on sMOX radionuclide release fractions and because of the conservatism in the ranges of values selected. Note the value used for the C-14 unbound fraction is assumed a constant value.
Activation Energy of Commercial SNF and sMOX (kJ/mol)	34,300	Because the dissolution rate of commercial SNF is assumed to be bound that of sMOX, the activation energy of sMOX was also assumed to have a same value to be consistent with the activation energy of commercial SNF in Leslie et al. (2007).
Activation Energy of HLW glass (kJ/mol)	40,000	The activation energy of 40 kJ/mol was back calculated using the dissolution rate data in the neutral pH (pHs of 7, 8, and 9) at 25 and 90 °C [77 and 914 °F] (Bechtel SAIC Company, 2004).

IV. PRELIMINARY β -SOAR OUTPUT

Example dose calculation results for commercial SNF and HLW glass using the β -SOAR are presented in Fig. 1 and 2, respectively. A 10 cm thickness of carbon steel was used as the waste package material and the redox condition was assumed to be reducing. The release results are comparable with those reported in the literature for other reducing environments (e.g., Marivoet et al., 2001). Actinide releases are not shown because their solubilities are low in the reducing environment.

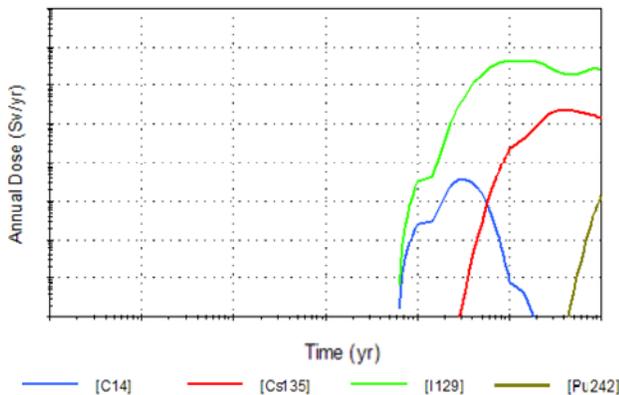


Fig. 1. Example β -SOAR dose results for only commercial SNF using combined degradation rate in a stylized reducing geologic disposal system.

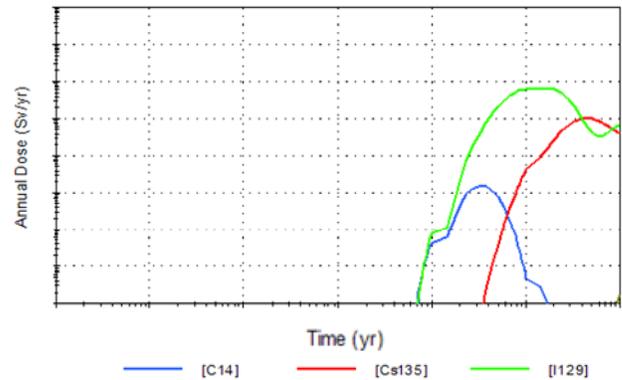


Fig. 2. Example β -SOAR dose results for only HLW glass using combined degradation rate in a stylized reducing geologic disposal system.

V. CONCLUSIONS

The β -SOAR considers two distinct waste forms, spent nuclear fuel (SNF) and high-level waste (HLW) glass associated with reprocessed SNF. The SNF separates commercial SNF from sMOX. An initial inventory contains 15 radionuclides (i.e., Pu-238, 239, 240, 242, U-232, 233, 234, 235, 236, 238, Np-237, C-14, Cs-135, I-129, Tc-99) bound on the SNF (or glass) matrix. Each bound radionuclide inventory has its own

dissolution rate under reducing conditions, oxidizing conditions or mixed condition. In addition to the bound radionuclide inventories, the SNF radionuclides are further segregated into unbound inventories for C-14, Cs-135, I-129, and Tc-99. These unbound inventories are rapidly released from the SNF waste forms upon contact with water and are used to conservatively model release of these radionuclides through other mechanisms (i.e. gap and grain inventories). These 15 different radionuclides from both the bound and the unbound inventories are modeled together in the present model. Each waste package contains a mixed (blended) waste form where the user may specify the ratio of commercial SNF, HLW glass and sMOX.

The β -SOAR has flexibility to be run primarily for different types of waste forms, different types of environments including redox conditions (e.g., oxygen or radiolysis effects), various waste loading, and more radionuclide types. A user defined waste form is available and can be modified for defining specific radionuclide inventories and fractional mobilization rates. Additionally, the user may modify the fractional rate of waste form mobilization rate by means of a multiplier. This parameter could be used to approximate mostly inhibiting effects on the waste form mobilization. For example, there could be protection by partially failed cladding, or redox variation or radionuclide sorption with iron or iron compounds. The user may also specify a fraction of the initial radionuclide inventory that is set aside and not available for mobilization (e.g., simulates intact cladding). The effects of temperature on the fractional mobilization rate can also be modeled using an Arrhenius relation for commercial SNF and HLW glass.

DISCLAIMER

This is a joint US NRC staff and CNWRA paper, and views expressed herein are preliminary and do not constitute a final judgment or determination of the matters addressed or of the acceptability of any licensing action that may be under consideration at the US NRC.

REFERENCES

1. Bechtel SAIC Company, "Defense HLW Glass Degradation Model." ANL-EBS-MD-000016. Rev. 02. ACN 001, ERD 001, ERD, 002. Las Vegas, Nevada: Bechtel SAIC Company, LLC (2004).
2. Broczkowski, J. J. Noël, and D. W. Smith, "The Inhibiting Effects of Hydrogen on the Corrosion of Uranium Dioxide under nuclear Waste Disposal Conditions," *Journal of Nuclear Materials*, **346**,16 (2005).
3. DOE, "Yucca Mountain Repository License Application," DOE/RW-0573, Rev. 0, ML081560400, Las Vegas, Nevada: DOE, Office of Civilian Radioactive Waste Management (2008).
4. DOE, "Review of DOE Spent Nuclear Fuel Release Rate Test Results," DOE/SNF/REP-073, Rev. 0, Idaho Falls, Idaho: Idaho National Engineering and Environmental Laboratory (2003).
5. C. Ferry et al., "Synthesis on the Spent Fuel Long Term Evolution," CEA-R-6084, Saclay, France: Commissariat A L'energie Atomique (2005).
6. V. Jain, , G. Cragolino, and L. Howard, "A Review Report on High Burnup Spent Nuclear Fuel-Disposal Issues," ML043020321. San Antonio, Texas: CNWRA (2004).
7. B. Leslie, , C. Grossman, and J. Durham "Total-System Performance Assessment (TPA) Version 5.1 Module Description and User Guide," ML072710060, San Antonio, Texas: CNWRA (2007).
8. J. X. Marivoet, D. Sillen, D. Mallants, and P. De Preter, "Performance Assessment of Geological Disposal of High-Level radioactive Waste in a Plastic Clay Formation," *Scientific Basis for Nuclear Waste Management XXIV*. B. P. McGrail and G. A. Cragolino (eds.), Materials Research Society Symposium Proceedings, Vol. 713, Paper No. J.J4-4, Warrendale, Pennsylvania: Materials Research Society (2001).
9. Nuclear Regulatory Commission (NRC "Dissolution Kinetics of Commercial Spent Nuclear Fuels in the Potential Yucca Mountain Repository Environment," NUREG-1914, ML 083120074, Washington, DC: NRC (2008).
10. NRC and The Center for Nuclear Waste Regulatory Analyses (CNWRA), "Performance Assessment: A Model for Scoping of Options and Analyzing Risk, Beta Version (β -SOAR)," User Guide, ML102720755, Washington, DC: NRC (2010).
11. Sandia National Laboratories (SNL), "MOX Spent Nuclear Fuel and LaBS Glass for TSPA-LA," ANL-WIS-MD-000022 REV 01, Las Vegas, Nevada (2007a).
12. SNL, "In-Package Chemistry Abstraction," ANL-EBS-MD-000037 REV 04 AD 01, Las Vegas, Nevada: Sandia National Laboratories (2007b).
13. D. W. Shoesmith, "Fuel Corrosion Processes under Waste Disposal Conditions." *Journal of Nuclear Materials*, **282**, 1(2000).
14. S. Sunder and D. W. Shoesmith, "Chemistry of UO₂ Fuel Dissolution in Relation to the Disposal of Used Nuclear Fuel," AECL-10395, Pinawa, Canada: Atomic Energy of Canada Limited (1991).
15. B. Rosborg and L. Wermer, "The Swedish Nuclear Waste program and the Long-Term Corrosion Behavior of Copper," *Journal of Nuclear Materials*, **379**, 142 (2008).

16. P. Van Iseghem, "Geological Disposal of Radioactive Waste – the Challenge of Assessing the Long-term Safety," SCK•CEN for ISL Sheffield, May 17, 2007, MOL, Belgium: SCK•CEN (2007).
17. C. N. Wilson, and W. J. Gray, "Measurement of Soluble Nuclide Dissolution Rates from Spent Fuel." *Scientific Basis for Nuclear Waste Management XII*.

V. M. Oversby and P. W. Brown (eds.). *Materials Research Society Symposium Proceedings*. Vol. 176. Warrendale, Pennsylvania: Materials Research Society. pp. 489-498 (1990).