

## **Appendix B**

### **Release Data for Initial Assessment Performed with the System Assessment Capability (Revision 0)**

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#### **1.0 INTRODUCTION**

The objective of this appendix is to present the data to be used in the initial assessment performed with the System Assessment Capability (SAC), Revision 0. This document also describes the basis for data selection and application. The release conceptual model for SAC is summarized in Appendix B: Preliminary System Assessment Capability Concepts for Architecture, Platform, and Data Management which can be found at <http://www.bhi-erc.com/vadose/sac.htm#info>. The initial assessment for Hanford waste source terms using the various release models, release software requirements specifications and code design, and test cases are described in (Kincaid et. al., 2000). This document can be found on the above mentioned web page.

#### **2.0 BACKGROUND**

Release model formulations for SAC Rev. 0 were from those found in Appendix D of the 200 Area composite analysis report (Kincaid et al, 1998). Models to be applied in the SAC Rev.0 assessment include the liquid, soil-debris, cement, salt-cake and reactor block models. Waste sources requiring the application of the glass and reactor compartment models were excluded because of their negligible influence on the overall output of the initial assessment where it is limited to approximately 1,000 years (i.e., 1944 to 3050). The technical basis for their exclusion has been documented (see Attachments A and B).

Each model is run on a 450 Mhz PC (Pentium III) with 128 Mbytes of RAM. Each run (VADER) requires between 3 and 8 seconds to complete. A capability to save and display a variety of different contaminant release representations has been developed to support a broad range of contaminant release analysis (see Independent Testing of Release Models for SAC Rev. 0 Runs at <http://www.bhi-erc.com/vadose/sac.htm#info>).

#### **3.0 INTERACTION WITH OTHER SAC MODULES**

The vadose zone release (VADER) program written in FORTRAN 95 receives the contaminant inventories output by the Inventory module and calculates release quantities. For a single realization of inventory of each contaminant, it calculates annual contaminant releases into the vadose zone over the specified time period of the initial assessment (i.e., from 1944 through 3050) according to specified release models (i.e.,

liquid, salt cake, cement, reactor block, and soil-debris) and single-chain radioactive decay. Releases involving liquid discharges pass directly through VADER from the inventory module to the vadose zone module. The program calculates the balance of inventory remaining at a site on an annual basis. VADER also accounts for projected waste transfers resulting from remediation efforts. VADER writes the release rate data in curies/year or kilograms/year into files to be read by the next processing step, the vadose zone flow and transport module (STOMP). Information flow is captured essentially in the SAC logic diagram present in Figure 5-1 of Kincaid et al., 2000.

A separate execution of VADER is required for each stochastic realization for each contaminant at each site. Stochastic variability of parameters in the release models are generated by the environment stochastic preprocessor (ESP) and passed to VADER via input keyword files.

#### 4.0 DATA GATHERED

In this section, parameters constituting the mathematical formulations for each release model are described along with the values for these parameters to be applied to the initial assessment. Table 1 summarizes the release models that are to be applied to the corresponding waste site types represented in the initial assessment. Specific conditions for modeling some of the sites are described in the footnotes. Data sets are applied site specifically or to aggregated sites. The release models will be applied to greater than 700 individual waste disposal and discharge sites during implementation of the initial assessment.

#### 4.1 Salt-Cake Model

The contaminant release mechanism of the salt-cake model is the dissolution of the structural matrix. As the matrix dissolves, all the contaminants are assumed to leach congruently at the same rate. When applied to the Hanford high-level waste tanks, the term "salt-cake" applies to the salt-cake, sludge, and hard heel residual in the tanks, which compose the "structural matrix." The release rate for a given contaminant is given by:

$$dM / dt = M_o A Q_w C_{wo}^{sol} / M_{wo} \quad (\text{after Equation D.48, p. D.14, Kincaid et al., 1998})$$

In which:

- $M_{wo}$  is the original mass of salt cake (kg).  $M_{wo}$  may also be derived by the product of tank waste volume and waste density.
- $M_o$  is the original quantity of contaminant in Ci or kg embedded in the salt-cake.
- $M = M(t)$  is the current quantity of the contaminant contained in the salt-cake (Ci or kg) at time  $t$
- $A$  is the surface area of salt-cake exposed to the release mechanism ( $\text{cm}^2$ )

- $C_{wo}^{sol}$  is the aqueous solubility of the salt-cake simulated as a nitrate salt (g/cm<sup>3</sup>)
- $Q_w$  is the site recharge rate in cm/yr, also termed "infiltration rate."
- $dM/dt$  is the rate of loss of contaminant from the salt-cake waste form per unit time  $t$  (the rate at which the contaminant enters the environment).

In the initial assessment, a value of 0.36 g/cm<sup>3</sup> for salt-cake solubility is applied to all high-level waste tank solids. The value is based on the highest nitrate concentration found in high-level waste tank drainable liquors (Serne and Wood, 1990). The same value was used in the application of release modeling in the tank waste remediation system final environmental impact statement (DOE 1996). The model takes the original tank waste volume(s) and converts it to original mass using a tank solid waste density of 1.58 ±0.2 g/ml. This value was obtained from the analysis of 525 core samples collected from the 177 high-level waste tanks (Chen, et al., 1998). Waste volumes, contaminant quantities within the waste volumes are provided by the inventory module. Sequences of recharge rates and tank cross sectional areas used by the salt-cake model are those used by the vadose zone module.

## 4.2 Cement Model

The contaminant release mechanism of the cement model is diffusion through the solidified waste material to the outer surface of the waste form. The rate of loss of contaminant for a given contaminant is given by:

$$dM / dt = -M_o (A / V) \sqrt{D / \pi t} \quad (\text{Equation D.61, p. D.17, Kincaid et al., 1998})$$

In which:

- $M_0$  is the original quantity of the contaminant contained in the cement (Ci or kg)  
This can be seen as a function of concentration (kg/cm<sup>3</sup> or Ci/cm<sup>3</sup>) and volume (cm<sup>3</sup>)
- $M$  is current quantity of the contaminant contained in the cement (Ci or kg)
- $A$  is the surface area of the cement structure (cm<sup>2</sup>)
- $V$  is the volume of the cement structure (cm<sup>3</sup>)
- $D$  is the diffusion coefficient of the contaminant (cm<sup>2</sup>/yr)
- $t$  is the elapsed time (years) from the beginning of release from containment
- $dM/dt$  is the rate of loss of contaminant from the cement waste form

In the initial assessment, the original quantity of contaminant contained in the cement is provided by the inventory module. Waste source dimensions (see Table 1 for waste source types) to calculate source volumes and surface areas were obtained primarily from the WIDS database. When source dimension information was missing, the source with missing data was compared to a source of similar type to obtain or calculate dimensions from which volumes and surface areas could be estimated. For each site or aggregated site, the surface area to volume ratio was assumed to be constant for all time periods

during any given realization of the assessment run. The results of this exercise are summarized in Table 2 for sources requiring the cement model and in the SAC Rev. 0 inventory database for all sites in the initial assessment.

Diffusion coefficients were obtained from several sources. Iodine-129 ( $3.5 \times 10^{-5} \text{ cm}^2/\text{yr}$ ) and technetium-99 ( $1.58 \times 10^{-4} \text{ cm}^2/\text{y}$  to  $1.89 \times 10^{-3} \text{ cm}^2/\text{y}$ ) diffusion coefficients were obtained from recent laboratory experimental work (Mattigod et al., 2000). Tritium ( $1.58 \times 10^0 \text{ cm}^2/\text{yr}$ ), uranium-238 ( $3.15 \times 10^{-5} \text{ cm}^2/\text{yr}$ ), strontium-90 ( $1.58 \times 10^{-3} \text{ cm}^2/\text{yr}$ ), plutonium-239 or 240 ( $1.58 \times 10^{-3} \text{ cm}^2/\text{yr}$ ) and cesium-137 ( $1.58 \times 10^{-2} \text{ cm}^2/\text{yr}$ ) were default values (Serne et al, 1992) and used in the 200 Area Composite Analysis assessment (Kincaid et al, 1998) or the waste isolation pilot plant no action alternative assessment (Buck et al, 1996). A default value was a value recommended to be used when specific data were lacking.

### 4.3 Reactor-Block Model

The contaminant release mechanism of the reactor-block model is leaching contaminants from the graphite blocks of the production reactors. The rate of loss of contaminant for a given contaminant is given by:

$$dM / dt = -M_o F_{rrr} \quad (\text{Equation D.65, p. D.19, Kincaid et al., 1998})$$

In which:

- $M_o$  is the initial quantity in Ci or kg of contaminant in the graphite core
- $F_{rrr}$  is the fractional release rate in  $\text{yr}^{-1}$ . Analyte specific, and ranges from 0 to 1.

This model, described originally in the surplus production reactor EIS (DOE 1989), generates a family of curves such that the smaller the value of  $F_{rrr}$ , the more elapsed time is required until a specific contaminant inventory is completely depleted from the graphite block.  $M_o$  serves as a multiplier or scaler.

The mass of individual contaminants in the graphite cores are provided by the inventory module and originated in the surplus production reactor EIS (DOE 1989). Fractional release rates for the SAC Rev.0 radionuclides found in reactor cores were tritium:  $3.65 \times 10^{-4}/\text{y}$ ; strontium-90:  $1.10 \times 10^{-2}/\text{yr}$ ; cesium-137:  $3.5 \times 10^{-5}/\text{yr}$ ; and plutonium-239:  $3.5 \times 10^{-5}/\text{yr}$  (DOE 1989). The values were experimental values corrected for the volume to surface area ratio of Hanford production reactors. The physico-chemical mechanisms governing leaching of radionuclides from graphite are unknown (White et al, 1984).

### 4.4 Soil-Debris Model

The rate of loss of contaminant for a given contaminant by the soil-debris model is given by:

$$dM / dt = -Q_w A C_w \quad (\text{Equation D.35, p. D.11, Kincaid et al., 1998})$$

Where:

$$C_w = C_{sol} \quad \text{in equation D.35 when the release process is solubility-controlled}$$

$$C_w = M / (\theta R A h) \quad \text{in equation D.35 when the release process is desorption-controlled where:}$$

$$R = 1 + (\beta K_d) / \theta$$

Switching régimes is controlled by comparing the remaining mass with the maximum mass  $M_{max}$  consistent with an aqueous phase saturated with the contaminant. If  $M$ , the mass remaining in the waste form is larger than the quantity  $M_{max}$  where :

$$M_{max} = \theta R C_{sol} A h$$

the release process is considered to be solubility controlled; otherwise it is desorption controlled

Definitions:

- $M_{max}$  is the maximum amount of contaminant possible in the source zone (in Ci or kg) without a precipitated phase.
- $M = M(t)$  is current quantity of contaminant contained in the source zone (Ci or kg)
- $Q_w$  is recharge rate for the site in cm/yr.  $Q_w$  can be considered constant, or considered to be time-dependent based on site climate and remediation activities.
- $A$  is surface area of the soil waste form exposed to the release mechanism ( $\text{cm}^2$ )
- $h$  is depth of the soil waste form the site in cm.
- $C_w$  is a coefficient expressing the effective release of the contaminant ( $\text{Ci}/\text{cm}^3$  or  $\text{kg}/\text{cm}^3$ )
- $C_{sol}$  expresses aqueous solubility of the contaminant in  $\text{Ci}/\text{cm}^3$  or  $\text{kg}/\text{cm}^3$ )
- $R$  is either a retardation factor or a soil apportionment factor (unitless) which depends on several factors:
  - $\beta$  Soil bulk density in  $\text{g}/\text{cm}^3$
  - $K_d$  Sorption factor ( $\text{cm}^3/\text{g}$ )
  - $\theta$  Soil volumetric content of water in soil (unitless fraction)
- $dM/dt$  is the rate of loss of contaminant from the source zone (the rate contaminant crosses the soil waste form boundary and enters the environment)
- $t$  is the elapsed time (years) from the beginning of release from containment

For the initial assessment, contaminant release from all burial grounds was simulated using the soil-debris and/or cement models. This decision was made recognizing that a number of the burial grounds contain stabilized waste (e.g., technetium, iodine, and uranium in high integrity concrete vaults or other form of concrete waste). Accounting in this manner maintains consistency with past low-level waste performance assessments (Wood et al., 1995; Wood et al., 1996).

For SAC Rev. 0, contaminant inventory in site source zones or aggregated areas is provided by the inventory module. Source zone depth/height information is also needed for all burial ground sites to run the soil-debris model. Some depth/height information was obtained from the WIDS database. An average depth/height was calculated based on the available WIDS data (5.349 m) and applied to all burial ground sites that did not have a depth/height value. The value was also applied to aggregated burial ground sites. In a few instances, cross sectional area values were not available in which case, a value of  $999\text{m}^2$  was applied (Table 3).

The soil-debris model is also applied to laboratory, storage and stack-type waste sources. Available dimensional data on these sites were obtained from the WIDS database. When waste source dimension information was missing, the waste source with missing data was compared to waste sources where dimension data were available to obtain or calculate/assign dimensions from which depths/heights and cross sectional areas could be estimated. The results of this exercise for laboratory, storage and stack-type waste sources are summarized in Table 4.

The soil-debris model is also applied to some of the unplanned release sites. Where cross-sectional area and depth/height information was missing, values of  $0.999\text{m}^2$  and  $0.999\text{ m}$  were assigned for cross-sectional area and depth/height, respectively.

Values for bulk density and volumetric moisture content for the 100, 200 and 300 Areas were calculated from available data (Peterson et al, 1996; Schalla et al, 1988, and Fayer et al, 1999) at depths no greater than 20 ft (i.e., the approximate maximum depth observed for burial grounds on the Hanford site). Those data are summarized in Table 5.

Site recharge rates, site cross-sectional areas and sorption factors are those used by the vadose zone module. In the case of the sorption factors, the soil-debris model uses the high-impact values associated with the vadose zone nearest the disposal facility (Table 6). The most commonly used  $K_d$  category is that associated with sites that are low organic, low salts, and near neutral pH in chemistry (i.e., Table 6, High-Impact F1).

For radionuclides where no specific solubility values are available, the aqueous solubility was fixed at a arbitrarily high default value ( $1 \times 10^{10}\text{ mg/L}$ ) so that the soil-debris release model would automatically select algorithms for sorption ( $K_d$ ) control in these cases (Kincaid, 1998). Radionuclides that were assigned default values<sup>1</sup> were tritium ( $9.7 \times 10^7\text{Ci/cm}^3$ ), technetium ( $1.7 \times 10^2\text{Ci/cm}^3$ ), iodine ( $1.77 \times 10^0\text{Ci/cm}^3$ ), strontium ( $1.37 \times$

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<sup>1</sup> The solubility (in  $\text{Ci/cm}^3$ ) for radionuclides requiring the high solubility default value ( $1 \times 10^{10}\text{ mg/L}$ ) was calculate according the procedure described in Kincaid et al., 1998, Appendix D, page D.21.

$10^6 \text{Ci/cm}^3$ ), and cesium ( $8.67 \times 10^5 \text{Ci/cm}^3$ ). Uranium ( $2.95 \times 10^{-11} \text{Ci/cm}^3$ ) and plutonium ( $3.58 \times 10^{-10} \text{Ci/cm}^3$ ) solubilities were calculated starting from values of uranium and plutonium estimated in Hanford groundwater (Wood et al, 1995). Solid controlling uranium and plutonium phases were assumed to be  $\text{UO}_2(\text{OH})_2 \cdot \text{H}_2\text{O}$  and  $\text{PuO}_2 \cdot \text{H}_2\text{O}$  as the basis for these calculations. Solubility for carbon tetrachloride is  $8 \times 10^{-4} \text{g/cm}^3$  (Rohay, 2000). The solubility of chromic anhydride ( $\text{CrO}_3$ ) ( $0.617 \text{g/cm}^3$ ) was used to represent the solubility of  $\text{Cr}^{+6}$  (Lange's Handbook of Chemistry, 1999).

#### 4.5 Statistical Treatment of Data

Based on available published information, data used for the various release model parameters were assigned the following statistical treatments.

- Salt cake Model
  - Solid Phase Solubility: Deterministic
  - Solid Waste Density: Mean/standard deviation, normal distribution
- Cement Model
  - Waste Surface Area: Deterministic
  - Waste Volume: Deterministic
  - Diffusion Coefficient: Deterministic with the exception of technetium-99 which is uniform with lower and upper limit.
- Reactor Block Model:
  - Fractional Release Rate: Deterministic
- Soil Debris Model
  - Distribution Coefficient ( $K_d$ ): Deterministic for tritium, technetium and chromium; log normal base e (defined by median and minimum values) for the other constituents (iodine, uranium, strontium, cesium, plutonium, carbon tetrachloride)
  - Solubility: Deterministic
  - Cross Sectional Area: Deterministic
  - Depth: Deterministic
  - Bulk Density: Mean/standard deviation, normal distribution
  - Volumetric Moisture Content: Mean/standard deviation, normal distribution
  - Recharge Rate: Deterministic

#### 5.0 ASSESSMENT DATA

Four spread sheets (100 Area, 200 West Area, 200 East/North Areas and the 300 Area) have been prepared in which the release model assessment data described in Sections 4.1 through 4.4 have been assigned to individual waste sites. The files containing this information can be found at <http://www.bhi-erc.com/vadose/sac.htm#info>.

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## Attachment A

### Determination of the Need to Assess the Effects of Reactor Compartment Source Term as Part of the SAC. Rev. 0

#### Summary

A review of the potential release of chromium and technetium-99 from reactor compartments undergoing disposal at the Hanford site's 218-E-12B burial ground (trench 94) was performed. The large inventory of three types of stainless steels making up the reactor compartments will corrode over time. The corrosion process in alkaline Hanford soils will convert the large inventory of elemental chromium to immobile  $\text{Cr}^3$  species (e.g.,  $\text{Cr}(\text{OH})_3$ ). The presence of a combined quantity of elemental iron and other iron species (e.g., magnetite, siderite) at a level almost 10-fold that of the chromium inventory will ensure a highly oxygen-depleted environment. The highly reduced environment assures that the chromium will remain in the immobile +3 state and not subject to migration in the vadose zone. A small amount of chromate (+6 state) contained within each reactor vessel (0.005% of total chromium inventory) would not be available for release within the first 1,000 years. When released, the mobile chromate would be reduced to immobile +3 state when it encountered the large inventory of elemental iron and other reduced iron species. In order for the inventory of technetium-99 (2.2 to 6.6 curies as pertechnetate) to be available for release requires corrosive breaching of the external compartment shell and the shell of the internal reactor vessel. Assuming the most conservative corrosion rate, it would take 1,250 years to breach the external reactor compartment shell. In addition, the highly reducing conditions created by the elemental iron will rapidly reduce the mobile pertechnetate to the highly immobile  $\text{TcO}_2$ . Both the technetium and the chromium will remain reduced and immobile until the iron has been completely oxidized. This would require a very long time due to the combination of the huge mass of iron to be reduced, a low infiltration rate of water and a relatively low rate of diffusion of oxygen through the vadose zone. With the length of initial assessment simulations set at 1,000 years, it is concluded that simulations of the reactor compartment source terms for chromium and technetium-99 will not be required for SAC Rev. 0.

#### Assessment

The groundwater /vadose zone integration project assessment design for the systems assessment capability (SAC) lists 8 radionuclides and 2 chemicals to be investigated as part of the SAC Rev. 0 assessment to begin in FY 2001<sup>2</sup>. Of those 10 constituents, two (chromium and technetium-99) make up part of the inventory of reactor compartments undergoing disposal at the 218-E-12B burial ground. The current plan is to dispose of 220 reactor compartments by expanding the current trench or constructing a second trench adjacent to the current trench. When completed, the trench(s) will be backfilled with soil and covered with an engineered barrier. At site closure, the steel inventory for

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<sup>2</sup> Aaberg, R.L., R.W. Bryce, A.L. Bunn, P.W. Eslinger, C.T. Kincaid, T.B. Miley, W.E. Nichols, M.C. Richmond and S.F. Snyder. "Groundwater/Vadose Zone Integration Project Assessment Design for the Systems Assessment Capability, Rev. 0 Draft, February 2000.

all disposed reactor compartments is estimated at 50,380,000 kg<sup>3</sup>. The elemental composition of those steels are summarized in Table 1.

Table 1: Elemental Composition of Reactor Compartment Steels

Steel Type <sup>4</sup>	Elements (% Composition)					
	Fe <sup>5</sup>	Cr	Ni	Mn	Si	C
304 Stainless	69.0	18-20	8-10.5	2.0	1.0	
Inconel 600	7.0	14-17	76.0			
HY-80	92.6	1.5	3.4	1.5		1.0

### Chromium Inventory, Speciation and Potential for Subsurface Migration

The chemical forms of chromium present are elemental and Cr<sup>+6</sup>. Elemental chromium is associated with three steels that make up the structure of the disposed reactor compartments and account for the overwhelming majority of the chromium inventory (4,709,100 kg)<sup>2</sup>. A small amount of chromate solution (1 kg/reactor compartment or 220 kg total) was used on board as a corrosion inhibitor and constitutes the remainder of the inventory. In the post closure site, chromium will be slowly released (over thousands of years) to the surrounding alkaline unsaturated soil in the form of immobile Cr<sup>+3</sup> (e.g., CrOH<sub>3</sub>) species as a result of corrosion of the reactor compartment steels. As a result of the low solubility of CrOH<sub>3</sub>, this release will occur at concentrations that are below the drinking water standard. Also released will be huge quantities of reduced iron (e.g., as

<sup>3</sup> Rhoads et al (PNL-9791, May 1994) estimated the average reactor compartment to contain 121,000 kg of HY-80 stainless steel, 41,000 kg of Type 304 stainless steel, and 67,000 kg of Inconel 600 steel. The plan is for 220 reactor cores to be disposed at the 218-E-12B Burial Ground (DOE 1996). Therefore, the total estimated inventory of elemental chromium and iron-containing steel in the burial ground is 220 times these estimates (i.e., 26,620,000 kg, 9,020,000 kg, and 14,740,000 kg, respectively).

The elemental composition for each of the stainless steels is summarized in Table 1. For estimating chromium inventories, the high weight percent chromium values from Table 1 were used in the calculations below.

#### Estimated Total Elemental Chromium Inventory

$$26,620,000\text{kg} \times 0.015 + 9,020,000\text{kg} \times 0.2 + 14,740,000 \text{ kg} \times 0.17 = 4,709,100 \text{ kg}$$

#### Estimated Total Elemental Iron Inventory

$$26,620,000 \text{ kg} \times 0.926 + 9,020,000 \text{ kg} \times 0.69 + 14,740,000 \text{ kg} \times 0.07 = 31,905,720 \text{ kg}$$

#### Estimated Total Inventory of Cr<sup>+6</sup> in Reactor Compartments

It is estimated that each reactor compartment contains 1kg of chromium (as Cr<sup>+6</sup>) (DOE 1996). Therefore, the total inventory of Cr<sup>+6</sup> is estimated as 220 kg (220 reactor compartments X 1kg Cr<sup>+6</sup>/reactor compartment).

<sup>4</sup> Elemental data for Cr, Ni, Mn, Si and C from ASM Metals Handbook , Vols. 4 (1991) and 6 (1994). ASM International, Materials Park, OH.

<sup>5</sup> Determined by difference (100%-% Cr, Ni, Mn, Si, and C).

magnetite, siderite) from an elemental iron inventory estimated at 31,905,720 kg or almost 10 times the amount of chromium inventory present<sup>2</sup>. The presence of large quantities of reduced iron will result in the presence of a continuous oxygen depleted environment, making it virtually impossible for chromium present to be oxidized over time to the highly mobile +6 state until the iron has been completely oxidized<sup>6</sup>. This would require a very long time due to the combination of the huge mass of iron to be reduced, a low infiltration rate of water and a relatively low rate of diffusion of oxygen through the vadose zone. If oxidizing conditions were to occur at some distant time in the future, chromium would continue to remain reduced and immobile. It is known that Cr(OH)<sub>3</sub> does not generally re-oxidize in the environment. The small inventory of chromate solution would be required to interact with this corrosion layer where it would be rapidly reduced to immobile Cr<sup>+3</sup> species. Release to the environment would occur beyond a 1,000 year time frame (see below for technetium-99).

### **Technetium-99 Inventory and Potential for Release from Reactor Compartments**

It is estimated that from 0.01 to 0.03 curies of technetium-99 are associated with each disposed reactor compartment. For the anticipated disposal of 220 reactor compartments, this would constitute a total technetium-99 inventory in the range of 2.2 to 6.6 curies. Release of the technetium-99 inventory (as pertechnetate) requires first breaching of the reactor compartment external structure (0.75 in thick steel) followed by breaching of the reactor vessel structure (approximately 6 inches of steel). Assuming a progressive corrosion process and a maximum corrosion rate of 0.0006 in/yr, breach of the reactor compartment external shell would occur in 1,250 years and breach through the reactor vessel structure in 11,250 years. Thus, no release of technetium-99 would occur in the first 1,000 years after disposal. It was also estimated that less than 0.2 % (4,400 to 13,200 µCi) of this inventory would be released from the source term in the first 10,000 years of corrosion<sup>2</sup>.

The highly reducing conditions created by the elemental iron will rapidly reduce any released mobile pertechnetate to the highly immobile TcO<sub>2</sub>. Technetium will remain reduced and immobile until the iron has been completely oxidized. This would require a very long time due to the combination of the huge mass of iron to be reduced, a low infiltration rate of water and a relatively low rate of diffusion of oxygen through the vadose zone. If the reactor compartment area did become oxidizing, TcO<sub>2</sub> could reoxidize to the mobile pertechnetate; however, it is likely that the technetium will be dispersed within a cemented zone of Fe(OH)<sub>3</sub> and Cr(OH)<sub>3</sub> formed through the oxidation of the reactor compartments. This will significantly inhibit the transport of oxygen to the TcO<sub>2</sub> and the subsequent diffusion of pertechnetate from the oxidized reactor compartments. As a result, release of technetium (if it occurs at all) will take place at some very extended time beyond 1,250 years and at a release rate that is expected to occur very slowly through diffusion from the iron and chromium oxide cement.

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<sup>6</sup> DOE (1996) Final Environmental Impact Statement on the Disposal of Decommissioned, Defueled Cruiser, Ohio Class, and Los Angeles Class Naval Reactor Plants. DOE/EIS-0259, U.S. Department of the Navy, April, 1996.



## Attachment B

### Determination of the Need to Assess the Effects of ILAW as Part of the SAC Rev. 0

#### Summary

A review of the most recent ILAW performance assessment results along with previous findings of the Composite Analysis for Low-Level Waste Disposal in the 200 Area Plateau of the Hanford Site was performed. Results indicated that for a base analysis case assuming annual recharge rate of 4.2 mm/y, the ILAW release at 1,000 years was insignificant ( $<10^{-5}$  of the all pathways standard). The groundwater impacts were also not significant at 1,000 years for a point of compliance well located 100 meters down-gradient of the disposal facility (i.e., groundwater contaminant concentration predictions compared to EPA drinking water standards). With the length of initial assessment set at 1,000 years, simulations of the ILAW source term will not be required for SAC Rev. 0. However, results from sensitivity cases for the ILAW facility at recharge rates near 50 mm/y may need to be considered for future SAC revisions.

#### Assessment

The 1998 version of the Hanford Immobilized Low-Activity Tank Waste Performance Assessment<sup>7</sup> provided an analysis of the long-term environmental and health impacts of the onsite disposal of Hanford immobilized low-activity tank waste packages. A more recent report<sup>8</sup> updates the performance of the proposed disposal action given new data that have been collected since this last assessment. The results of the new assessment are based on the remote handled trench concept as the baseline for the ILAW Disposal Project<sup>9</sup>

In general, the current analysis<sup>2</sup> shows significantly less impacts than did the previous performance assessment. The main reason for the smaller impacts are improved information concerning waste form release, increased groundwater flow beneath the disposal facility, better understanding of the waste inventory, and new information about vadose zone retardation of important radionuclides. In the analysis, contaminant release data from a relevant glass form composition was used rather than draft procurement limits or contaminant release data from a higher temperature glass.

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<sup>7</sup> F.M. Mann, R.J. Puigh II, P.D. Rittmann, N.W. Kline, J. Voogd, Y.Chen, C.R. Eiholzer, C.T. Kincaid, B.P. McGrail, A.H. Lu, G.F. Williamson, N.R. Brown, and P.E. LaMont, Hanford Immobilized Low-Activity Tank Wte Performance Assessment, DOE/RL-97-69, U.S. Department of Energy, Richland, Washington, March 1998.

<sup>8</sup> F.M. Mann, S.H. Finfrock, E.J. Freeman, R.J. Puigh II, D.H. Bacon, M.P. Bergeron, B.P. McGrail, and S.K. Wurstner, White Paper Updating Conclusions of the 1998 ILAW Performance, DOE/ORP-2000-07, Rev.0, Fluor Federal Services, Richland, Washington, and Pacific Northwest National Laboratory, Richland, Washington, April 2000.

<sup>9</sup> W.J. Taylor, Contract No. DE-AC06-99RL14047-Decision to Change the Immobilized Low Activity Waste (ILAW) Disposal Baseline to Proceed with the Remote-Handled Trence Alternative, letter 99-DPD-066 (correspondence control number 9958849), Department of Energy, Richland, Washington, December 1, 1999.

For the DOE time of compliance (1,000 years), the all pathways dose from the key radionuclides, technetium-99 and iodine-129, was determined to be relatively insignificant ( $3 \times 10^{-5}$  mrem/yr vs 25.0 mrem/yr performance objective). The results were consistent with the results of “The Composite Analysis for Low-Level Waste Disposal in the 200 Area Plateau of the Hanford Site” that showed the Impact of ILAW disposal to be insignificant ( $<10^{-5}$  of the all pathways standard) during the DOE time of compliance of 1,000 years.<sup>10</sup> It is important to note, however, that this “base case” analysis considered a recharge rate of 4.2 mm/y only. Additional analyses at higher recharge rates, reaching 50 mm/y maximum are planned. The results from these simulations could result in dose impacts at 1000 y that may need to be considered in future SAC revisions.

The estimated groundwater impacts for the base analysis case were also shown to be insignificant at a point of compliance, a well located 100 meters downgradient of the disposal facility, when compared with EPA drinking water standards. Key radionuclides for determining peak groundwater concentrations were technetium-99 and iodine-129 (beta/photon emitters) and <sup>237</sup>Np and uranium (alpha emitters). For beta/photon emitters, estimated dose at 1,000 years was  $8 \times 10^{-6}$  mrem/yr as compared to a 4 mrem/yr performance objective. This dose is equivalent to an average caused by a predicted annual human consumption of groundwater containing concentrations of  $1.8 \times 10^{-3}$  pCi/L and  $2 \times 10^{-6}$  pCi/L of technetium-99 and iodine-129, respectively. For alpha emitters, estimated dose at 1,000 years was 0.0 mrem/yr as compared to a 15.0 mrem/yr performance objective. The higher  $K_d$  of neptunium and uranium results in much longer travel times through the vadose zone such that no impact to groundwater is expected in 1000y. Estimates for the impacts of other hazardous materials in the ILAW waste were also investigated. The resulting concentrations in the groundwater near the site were more than a factor of 5,000 less than the performance goals for these materials at 10,000 years after facility closure.

Uncertainties in ILAW waste form release were also investigated to a limited extent. Changing the water infiltration rate from 4.2 mm/yr to 0.9 mm/yr reduced the estimated impact by a factor of 10 for the remote handled trench. The release rate from a concrete vault design concept had significantly higher release rates from the waste form. The higher rates were attributed to a larger fraction of the glass exposed to higher pH leachate than in the trench calculations. Although the release rate from the concrete vault facility was a factor of 10 greater than the trench simulation, the estimated impacts are still approximately a factor of 40 below the performance objectives for this proposed disposal option. As noted above, calculations are underway at significantly higher assumed recharge rates. Inclusion of these results in future SAC revisions may be required. Finally, the Oregon Office of Energy has expressed concern about the impacts of organic acids (humic and fulvic) on the performance of ILAW glasses. Experimental data regarding the effects of these organic acids on ILAW glasses is not available and so the estimated impacts (if any) have not been assessed.

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<sup>10</sup> C.T. Kincaid, M.P. Bergeron, C.R. Cole, M.D. Freshley, N.L. Hassig, V.G. Johnson, D.I. Kaplan, R.J. Serne, G.P. Streile, D.L. Strenge, P.D. Thorne, L.W. Vail, G.A. Whyatt, S.K. Wurstner”, Composite Analysis of Low-Level Waste Disposal in the 200-Area Plateau of the Hanford Site, PNNL-11800, Pacific Northwest National Laboratory, Richland, Washington, March 1998.