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**APPENDIX B**  
**RELEASE CONCEPTUAL MODEL**

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## APPENDIX B

### Release Conceptual Model Abstract

The objective of this appendix is to describe the release conceptual model developed for use in the System Assessment Capability (SAC), Rev. 0.

Output from the release technical element will provide estimates of contaminant flux to the vadose zone from various waste sources on the Hanford Site to allow feasibility testing of the SAC (Rev. 0) and to complete an initial assessment of risk and impact from Hanford Site waste. The data from this element feed into the vadose zone technical element (Figure B-i).

The release technical element will take the results of the analyses from the inventory technical element in the form of spatial and temporal placement of waste inventory in waste sites and facilities and the physical and chemical characteristics of the contaminant inventory. The release model requires data that define the geometry of the release facility, the nature of waste containers, natural and artificial infiltration through the waste form, and processes controlling the breakdown of the waste and the release of radionuclides and chemicals of concern.

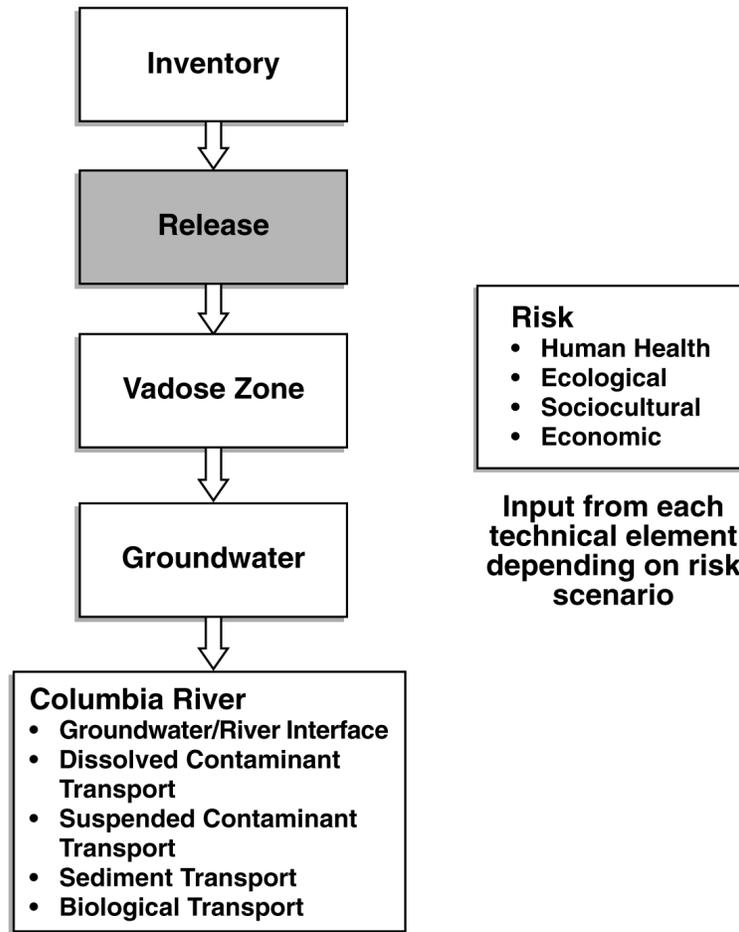
Information describing the geometry of the waste disposal facilities and the disposal containers is available through historic records for the various waste disposal sites. Natural infiltration rates have been estimated for surface locations on the Hanford Site from previous studies. Documentation of releases of water to the various waste sites is incomplete in site historic records. Information on partitioning coefficients, solubility, dissolution rates, corrosion rates, leach rates, diffusion coefficients, and decay and degradation rates will be obtained from published sources.

The release technical element will provide estimates of contaminant release from liquid discharges, solid waste, tank salt cake and sludge, glass waste, graphite cores, and reactor compartments. Estimates of contaminant flux for four classes of radionuclides and two chemicals of concern will be provided as input data to the vadose zone model element. The four mobility classes will range from those shown to be highly mobile to highly immobile under conditions observed at the Hanford Site.

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Figure B-i. System Assessment Capability System Conceptual Model.



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## APPENDIX B

### RELEASE CONCEPTUAL MODEL

#### B.1 INTRODUCTION AND BACKGROUND

This document describes a preferred alternative for the containment failure/release conceptual model to be used in Revision 0 of the System Assessment Capability (SAC) for determining the release of contaminants (i.e., radionuclides and chemicals) from selected waste source terms for the Hanford Site. The recommendation derives from an examination of (1) the present and future status of selected source terms; (2) release models that have been used in past assessments of these source terms; and (3) identification of technical elements and evaluation of options for selection of the preferred alternative. Uncertainty and assumptions on which the model is based, and outstanding issues influencing the enhancement of capabilities for the proposed conceptual model of release, will also be assessed in this white paper.

For this document, contaminant releases are considered for solid waste burial sites (e.g., low-level radioactive waste burial grounds, Environmental Restoration Disposal Facility [ERDF], other solid waste burial grounds), high-level waste tanks and associated retrieved and treated waste (immobilized low-activity waste [ILAW]), and special waste forms (i.e., submarine reactor compartments). Also considered are the anticipated disposal configurations for reactor blocks and 200 Area processing and storage facilities. Not included in this assessment are sources where contamination has breached the engineered structure and entered the vadose zone or groundwater. Examples of source terms that meet this definition include high volume liquid release sites, leaking single-shell tanks, and injection wells. For these types of examples, release is expected to be handled within the appropriate vadose zone or groundwater transport model. Also not considered were releases into the Columbia River from sources other than the Hanford Site, and where the Columbia River itself can be considered a source term (e.g., release of contaminants in river sediments to the water column). Releases to the atmosphere are also not considered.

Release models depend on the character of the source facility. A description of present or planned engineered source terms for which release models are addressed in this document are summarized in Attachment 1.

#### B.2 ASSESSMENT OF PAST AND PRESENT CONCEPTUAL MODELS OF RELEASE

Models were evaluated in the *Hanford High-Level Defense Waste Environmental Impact Statement* (HDWEIS) (DOE 1987) for use in simulating the release of contaminants from waste forms that might be disposed of in the Hanford 200 Areas Plateau. Release models consisted of four types: adsorption-controlled release, solubility-controlled release, linear release, and diffusion controlled release. For adsorption-controlled release, release was governed by the

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retardation factor and source quality of individual radionuclides in the solid phase. For solubility controlled release, radionuclides were carried away from their source at their maximum solution concentration. The linear release model was viewed as a model for dissolution-controlled release of a non-decaying chemical (e.g., nitrate). The diffusion controlled release model was considered applicable when a protective barrier was present, replacing advective transport of contaminants to the containment boundary with diffusion transport of contaminants to the containment boundary. This assumes that the surface barrier (or cover) can be shown to be so effective as to cause diffusion to dominate over advection in the region below the barrier; thus, the release is represented as a diffusion-dominated phenomena.

For the HDWEIS, model selection was made for specific waste forms, considering both the presence and absence of protective barriers (DOE 1987). For salt cake and sludge in single-shell tanks, without a protective barrier, the solubility model was used for radionuclides and the linear release model for non-decaying chemicals. For release of liquid from single-shell tanks, an adsorption model was used. For release of transuranic wastes (e.g., in low-level waste burial grounds), adsorption and solubility models were used. In the presence of a protective barrier, a diffusion controlled model was recommended for two of the above scenarios (salt/sludge, liquid release from tanks), with a decaying source for the radionuclides and a non-decaying source for chemicals.

For the *200 Area Composite Analysis* (Kincaid et al. 1998), five idealized generic types of contaminant source terms (i.e., generic waste form types) were considered for release of radionuclides: soil debris, saltcake/sludge waste, glass waste, cement waste, and reactor block waste. Release was conceptualized to occur as a result of water percolating through a well-mixed waste form, with radionuclides only being lost from the source term via radioactive decay. Releases of contaminants from a soil-debris waste form were proposed to be controlled primarily by partitioning between the aqueous and solid phases. If inventory levels in the waste form type were high enough, release was considered to be solubility controlled. For the cake type waste form, release of contaminant was at a constant rate congruent with the dissolving of a major structural component of the waste (e.g., salt or sludge in high-level waste tank). For the glass waste form, releases of contaminants occurred congruent with the surface dissolution of the glass.

The model took into account changes in dissolution rate as the size of the waste form shrank with time. For the cement waste form type, contaminants inside the waste form were assumed to diffuse toward the outer surface, where they were released into the infiltrating water flowing past the waste form. Therefore, release from this source term was controlled by the contaminants' effective diffusion coefficient in the waste form. For the reactor block waste form type, release of contaminants was assumed to be described by rates calculated from experimental leach test data.

Other more specific applications of release models are described below.

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### **B.2.1 Release from High-Level Waste Tanks**

The release of contaminants from tank residuals following retrieval of the high-level waste has been evaluated assuming that the structural integrity of the tanks degrade over time, allowing recharge water to enter the tank, dissolve contaminants from the residuals, and drain out into the surrounding vadose zone through cracks in the tank (Jacobs Engineering, Inc. 1999). It was assumed that essentially all the drainable liquid waste would be recovered from the tank, eliminating the potential for contaminant release for a period of time following waste retrieval and tank closure. Subsequently, infiltrating water would enter the tank. An enhanced RCRA Subtitle C barrier was assumed to be constructed over the tank farm following waste retrieval and tank stabilization.

The radiological and chemical source term consisted of the inventory of a 1% residual waste remaining in the tanks after sluicing. The approach to estimating the source term parameters for the residual tank waste assumed that, over time (following closure), the liquid containment integrity of the tanks degraded and the release of contaminants occurred from dissolution by infiltrating water that migrates into and out of the tanks through cracks. This approach was in agreement with the assumed mechanism for contaminant releases from single-shell tanks following closure (Serne and Wood 1990).

A constant concentration release model (analogous to solubility) was used to develop contaminant flux rates based on post-retrieval inventory data (COGEMA 1998). Specific waste type wash factors (Colton 1995, 1996) were assumed to provide representative concentration values for how contaminants would be dissolved in infiltrating water and released over approximately 30% of the tank base area. This assumption was based on engineering judgment and available data on potential leak mechanisms (WHC 1994). For tanks in general, the best estimate values for contaminant release rates from the tank residuals were based on the empirical solubility constraints using data that were most directly applicable to the waste type contained in the individual tanks.

### **B.2.2 Release from Immobilized Low Activity Waste**

The conceptual model for release of contaminants from the ILAW form glass and transfer out of the vault to the vadose zone was depicted as follows (Mann et al. 1998).

Infiltration of moisture from precipitation enters the engineered system. The water moves toward the waste form, but most of it is diverted by an intact capillary barrier. The water that is not diverted is chemically modified by the local environment, and interacts with the metal containers containing the waste form. Corrosion of the container occurs over time. Subsequently, the container is breached. Water containing corrosion products from the container as well as constituents from the surrounding soil interacts with the waste form. The waste form (silicate glass) corrodes in the following three phases, releasing radionuclides: (1) the glass reacts with water under dilute conditions to release components of the glass into solution; (2) corrosion rates approach a very low constant value as saturated fluid conditions are approached; and (3) secondary mineral phases may form from the saturated fluid resulting (with time) in an acceleration of the forward rate of release. The moisture containing the released

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contaminants travels downward through the vadose zone, until the contaminants reach the unconfined aquifer.

In the recent performance assessment for ILAW (Mann et al. 1998), dissolution of the waste form (glass) along with local chemical conditions was assumed to control the release rate of the radionuclide contaminants. Since the waste form was unspecified at the time, a time dependence of the radionuclide release rate was calculated using the non-technetium initial fractional release rate of 4.4 ppm/y used in all model simulations. The conceptual model for the performance and degradation of the waste package ignored the presence of the container. The waste form was assumed to be nearly cubical. The contaminants were assumed to be evenly distributed among waste packages, and contained minimal void space (no greater than 1%). The model provided estimates of contaminant release out to 10,000 years.

### B.2.3 Release from Solid Waste Burial Sites

**B.2.3.1 Release of Contaminants from Low-Level Waste Burial Grounds.** Release of contaminants from low-level waste cannot be modeled precisely because of the variability of the chemical and physical reactions that occur in the waste material. In the real system, radionuclides and chemicals are distributed in a heterogeneous fashion among different waste materials. Waste package containers fail at different rates because of the variability in waste material, and variable types and quantities of radionuclides and chemicals are dissolved into the infiltrating water over time--depending on which waste material contacts a particular volume of water. Therefore, averaging concepts are used in modeling that simplify the mathematical representation of the real system. These concepts must be justified as being a conservative representation of the real system.

Past performance assessments of low-level waste burial grounds (Wood et al. 1995, 1996) have used three release processes (advection dominated, diffusion dominated, and solubility limited) to address contaminant release from these waste systems. The advection-dominated release model (mixing-cell cascade model; Kovak et al. 1990) was used to simulate the processes of release from unstabilized (not contained) waste. In this case, the entire inventory was immediately available for release. Neither sorption effects nor decay were factored into these calculations. For unstabilized waste, the radionuclides exited the facility at a rate determined by the flow of water and the amount of dispersion (mixing in the disposal unit [i.e., by near-field transport processes]). The diffusion-dominated release model was used to simulate the release of contaminants from stabilized contained wastes. In the absence of convection through the waste container, the release was modeled as a diffusion-limited process. Release from the waste form was represented as a diffusion coefficient. In addition to the diffusion-dominated release of radionuclides from the burial trench, an alternative approach was to specify a solubility or corrosion control limit in the waste form. Infiltration rates of 5 cm/yr were chosen for the category 1 facility and 0.5 cm/yr for the category 3 facility for different modeling scenarios. Category 1 and category 3 wastes are distinguished by their radionuclide content, as indicated in DOE (1997a).

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Additional assumptions to introduce conservatism into the source-term release were made for conduct of performance assessments for 200 West and 200 East Area Low-Level Waste Burial Grounds (Wood et al. 1995, 1996):

- It was assumed that containers were not present in the disposal facility, and had no influence on the time at which waste would come in contact with infiltrating water. Distributed failure of different containers would contribute to the reduction of the peak radionuclide flux value.
- For waste materials disposed directly without treatment, it was assumed that the radionuclide inventory in those wastes were immediately available for release into the infiltrating solution (instantaneous dissolution).
- It was assumed that all the infiltrating water received the maximum amount of dissolved radionuclides prescribed by the release mechanism assumed in the modeling analysis.
- For those wastes that were incorporated into a waste form that controls radionuclide release by diffusion and sorption or solubility mechanisms (i.e., grout), it was assumed in the models that the diffusion coefficient values remain constant over time.

The conceptual model assumed additional conditions to facilitate numerical modeling:

- Radionuclide inventories were assumed to be homogeneously distributed among the wastes.
- One type of release mechanism was assumed. In addition to instantaneous dissolution and diffusion, a steady-state concentration was assumed as a separate type of release to simulate solubility control of radionuclide release.
- Unit concentrations (for solubility) or unit quantities (for instantaneous release and diffusion controlled release) of single radionuclides were assumed for the modeling runs.

In the assessment of the 200 West Area burial grounds (Wood et al. 1995), descriptions were provided for releases of radionuclides (i.e., carbon-14 and iodine) from waste forms (highly shielded containerized waste [activated metal] and grout) often found in low-level waste burial grounds.

**B.2.3.2 Release of Contaminants from ERDF.** Previous modeling for comparing alternative ERDF designs incorporated waste release mechanisms (e.g., waste dissolution and diffusional release) that were applied to vitrified and grouted waste. It was determined that, given the greater importance of vadose zone travel time, the advantage of accounting for diffusion through the liner was not warranted. Additional reasons given to ignore this mechanism included the computational difficulties in simulating diffusion as a plug flow process, and the lack of information regarding constituent-specific diffusion coefficients (WHC 1993).

In a more recent study (DOE 1994), contaminant releases from the ERDF were predicted, assuming the waste was untreated. In this case, constituent releases were viewed as being

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controlled by either solubility or partitioning between the waste and pore water. As infiltrating water percolated through the waste, it dissolved contaminants to form leachate. The contaminant concentrations in the leachate were controlled by the soil water partitioning unless the leachate concentration was predicted to exceed the constituent solubility, in which case the concentration was solubility limited. For simplicity, it was conservatively assumed that the leachate was released at time zero, with no decay. Furthermore, changes in solubility due to interactions with other waste constituents were not considered, and plug flow of mass through the liner was assumed.

### B.2.4 Release from Reactor Blocks

In the reactor release model, irradiated solids were assumed to release contaminants (via leaching of graphite and corrosion of activated metal) into infiltrating water over time. Release of lead (a shielding component in the reactor block system) was based on a solubility-controlled release into the infiltrating water. The reactor block release model was used to simulate release from each of the surplus reactors (DOE 1989). No credit was taken in the analyses for liner or leachate collection systems. For the dosimetric analysis, it was assumed that one-half of the released carbon-14 and other constituents were transported by the groundwater, and the remainder was assumed to be transported directly out of the burial site into the atmosphere. For the various scenarios, a post-disposal assessment period of 10,000 years was evaluated.

**B.2.4.1 Release Rates for Graphite.** A release function was developed for carbon-14 release from graphite (key reaction assumed to be carbon reacting with oxygen in the air dissolved in the water). A rate of  $2.2 \times 10^{-12}$  g/cm<sup>2</sup>/day was determined (wet storage conditions) (Gray 1982). The resulting release would extend over 23,000 years. White et al. (1984) studied the leaching of carbon-14 from demineralized water, and calculated a release rate of  $5.5 \times 10^{-7}$  cm/day at 22 °C, which is in good agreement with the results of Gray (1982). White et al. (1984) was able to develop an equation to describe the release rates from irradiated graphite in saturated groundwater as a function of time and temperature. Release rates under dry storage conditions were related to release rates under wet storage conditions, based on the relative humidity of the surrounding air. Carbon-14 release rates from irradiated metals are unknown. For activation products in metal components in the reactor block, release rates can be equated with corrosion rates (see below).

**B.2.4.2 Release Rates of Radionuclides (other than C-14) from Graphite.** Release rates of radionuclides other than carbon-14 have been reported by White et al. (1984) for tritium, cesium-134, cobalt-60, barium-133, iron-55, europium-154, and chlorine-36.

**B.2.4.3 Release Rates from Shielding and Metal Components.** The iron shielding and aluminum components of the reactor contain activation products that are subject to release as corrosion occurs. To provide a release rate for these radionuclides, iron was assumed to corrode at a rate of 5 mils (0.005 in./yr) or  $3.5 \times 10^{-5}$  cm/day. Based on this corrosion rate, the radionuclides would release over a period of 390 years (DOE 1989). Aluminum components were assumed to corrode at a rate of 0.1 mil/yr, yielding a release rate of  $7 \times 10^{-7}$  cm/day. Based on this corrosion rate, the radionuclides in the aluminum would release over a period of 1,250 years (DOE 1989).

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**B.2.4.4 Release Rates of Lead from Shielding.** Water was assumed to reach a solubility limited lead concentration of 0.29 mg/L. Lead migration would be very slow. Predicted peak concentrations would not occur for between 4.5 million and 45 million years for disposal in the 200 West Area (DOE 1989).

### B.2.5 Releases of Contaminants from Reactor Compartments

Radioactivity in the reactor compartment is primarily in the form of corrosion-resistant activated metals (i.e., nickel-63, carbon-14, niobium-94, nickel-59, selenium-79, and technetium-99) that make up the internal structure of the reactor pressure vessel. The Navy estimated that more than 99% of the inventory is found in these metals. Also present within the reactor vessel are polychlorinated biphenyls (PCBs) used for thermal shielding, and lead (used for radioactivity shielding). After 500 years, only nickel-63 remains. High-strength carbon steels and very high tensile strength alloyed steels form the exterior of reactor compartment disposal packages (DOE 1996).

A previous study (Rhoads et al. 1994) considered the disposal of a group of 120 reactor compartments at a 200 East Burial Ground (Attachment 1) as a potential nickel radionuclide source due to the presence of metal alloys inside the compartments that contain activated nickel (nickel-59 and nickel-63). The compartments were modeled with average quantities of nickel alloy and activated nickel, based on total inventories found in reactor compartments. Nickel radionuclides were modeled as activated constituents of corrosion resistant steel and steel alloys. Recharge (0.1, 0.5 and 6 cm/yr) passing through this area was assumed to contact the reactor compartment and exit saturated with nickel. Corrosion rates used were 0.0001 mg/cm<sup>2</sup>/yr and 0.0002 mg/cm<sup>2</sup>/yr, respectively, for the different corrosion resistant steels. It was found that nickel-63 would decay to negligible levels ( $1 \times 10^{-10}$  pCi/L) prior to reaching the aquifer, even under the postulated wetter condition (0.5 cm/yr versus 6 cm/yr infiltration rate).

In a similar study (Rhoads et. al 1992), the release and migration of lead from the reactor compartments was also estimated. As with nickel, average lead quantities were used. Lead was very conservatively assumed to be immediately available for dissolution, so that all groundwater contacting the 15.2 x 15.2 m<sup>2</sup> reactor compartments would exit the area being fully saturated with dissolved lead. Lead solubility was set at roughly double experimental results.

A solubility limited concentration of 15 ppb was used to simulate the release of PCBs from reactor compartments and to assess impacts to the Columbia River. Downstream concentrations of PCBs in the aquifer would be less than 0.5 ppb for the postulated wetter condition, and less than 0.1 ppb for the current climate (DOE 1996).

### B.2.6 Releases of Contaminants from Process Facilities (Canyons and Tunnels)

Published work on release of contaminants from process and storage facilities is unknown. Kincaid et al. (1998) chose to exclude such source terms from the 200 Area Composite Analysis based on 1) the absence of data on radionuclide inventories for these facilities; and 2) the fact that these facilities appear to retain excellent waste performance characteristics for stabilization of radionuclide contamination. For the latter point, it was assumed that it was unlikely that such

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facilities would be a significant source of groundwater contamination, especially in the next 1,000 years. Any assessment of release within the SAC will likely employ a soil-debris release model capability. Current plans for decommissioning key process and storage facilities on the Hanford Site are summarized in Attachment 1.

### B.3 CONCEPTUAL MODEL PROPOSAL OF RELEASE FOR THE SAC (REV. 0)

This section describes proposed conceptual models of release for the SAC (Rev. 0). The proposed conceptual models are identified from an evaluation of technical elements and options that depict release capability. Technical elements that describe the scope of release model capability were model type, containment strategy, inventory distribution, and recharge representation. Table B-1 summarizes possible options that could be implemented under each of these technical elements and highlights a combination of options that represents the preferred alternative for representing the proposed release conceptual model for the SAC (Rev. 0).

**Table B-1. Options for Identifying the SAC (Rev. 0) Alternative.**

Technical Elements			
MODEL TYPE	CONTAINMENT STRATEGY	INVENTORY DISTRIBUTION	RECHARGE REPRESENTATION
<b>Simple</b> <ul style="list-style-type: none"> <li>• Advective transport</li> <li>• Traditional geochemical parameters</li> </ul>	<b>Ignore Packaging</b> <ul style="list-style-type: none"> <li>• Contaminants released directly to infiltrating water</li> </ul>	<b>Homogeneous</b> <ul style="list-style-type: none"> <li>• Contaminants uniformly distributed throughout waste</li> </ul>	<b>Steady State</b> <ul style="list-style-type: none"> <li>• Averaged over long-term simulations</li> </ul>
<b>Semi-Complex</b> <ul style="list-style-type: none"> <li>• Distributed flow</li> <li>• Traditional geochemical parameters</li> </ul>	<b>Expert Judgment of Time of Failure</b> <ul style="list-style-type: none"> <li>• Design Criteria</li> </ul>	<b>Pseudo-Homogeneous</b> <ul style="list-style-type: none"> <li>• Distribute inventory</li> <li>• Multiple simulations</li> <li>• Sum outputs</li> </ul>	<b>Step-Wise Steady State</b> <ul style="list-style-type: none"> <li>• Effects of barrier failure</li> <li>• Long term climate change</li> </ul>
<b>Complex</b> <ul style="list-style-type: none"> <li>• Distributed flow</li> <li>• Geochemical competence (e.g., chemical speciation and complex interactions)</li> </ul>	<b>Explicit Process Models of Waste Package Degradation</b>	<b>Non-Homogeneous</b> <ul style="list-style-type: none"> <li>• Knowledge of spacial distribution</li> </ul>	<b>Fully Transient</b> <ul style="list-style-type: none"> <li>• Simulating short term changes (e.g., wet seasons that lead to non-linearity)</li> </ul>

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**SAC Rev. 0 Preferred Alternative**

The release model type can vary from simple to complex. On the simplistic side, release can be formulated as a mathematical expression of release controlled by traditional geochemical processes (i.e., sorption/desorption, solubility, dissolution, diffusion, corrosion, leaching). Contaminants are moved away from the source of release through advective transport. Such an

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approach was used in the implementation of the composite analysis (Kincaid et al. 1998), where the release models were implemented within the Composite Analysis.xls Excel<sup>TM</sup> workbook. A semi-complex model would couple simplified geochemistry with distributed flow. Distributed flow accounts for differential flow paths and flow rate influences by subsurface soil layering. Distributed flow around waste forms would result from the placement of engineered barriers over the waste sources. In the most complex option, models could be implemented that explicitly consider, for example (1) distributed flow; and (2) geochemical competence (e.g., geochemical speciation that may occur between the waste form and surrounding soil that could influence waste form release rates). Geochemical speciation has been incorporated into recent simulations of contaminant release from a borosilicate waste glass (Chen et al. 1997, Bacon and McGrail 1997).

Waste source containment results in delays in the length of time in which infiltrating water is able to contact waste forms and initiate reaction and release. Waste form containment can be addressed in several ways. First, one can choose to ignore all aspects of containment for all release simulations. Alternatively, one can invoke expert judgment with regard to the time of failure for different containment types (e.g., engineered barriers, vaults, tanks, steel canisters) based on (for example) waste package design criteria. A third option would be to provide explicit process models that depict waste containment degradation. Such models might rely on more environmentally relevant data than is provided by design criteria.

Inventory can be represented as homogeneous, pseudo-homogeneous, or non-homogeneous. In the homogeneous and most simplistic case, contaminants are assumed to be uniformly distributed throughout waste forms. A pseudo non-homogeneous case can also be considered if the waste inventory can be representatively distributed among the various waste forms present in the waste source. In this case, multiple simulations can be performed on each waste form type, and the outputs of the individual simulations summed as output to the vadose zone. One possible application of this option would be to a low-level waste burial grounds containing pre-1980 waste. In the third option the waste is considered non-homogeneous, in which case the spatial distribution of the waste within a given waste package is well known.

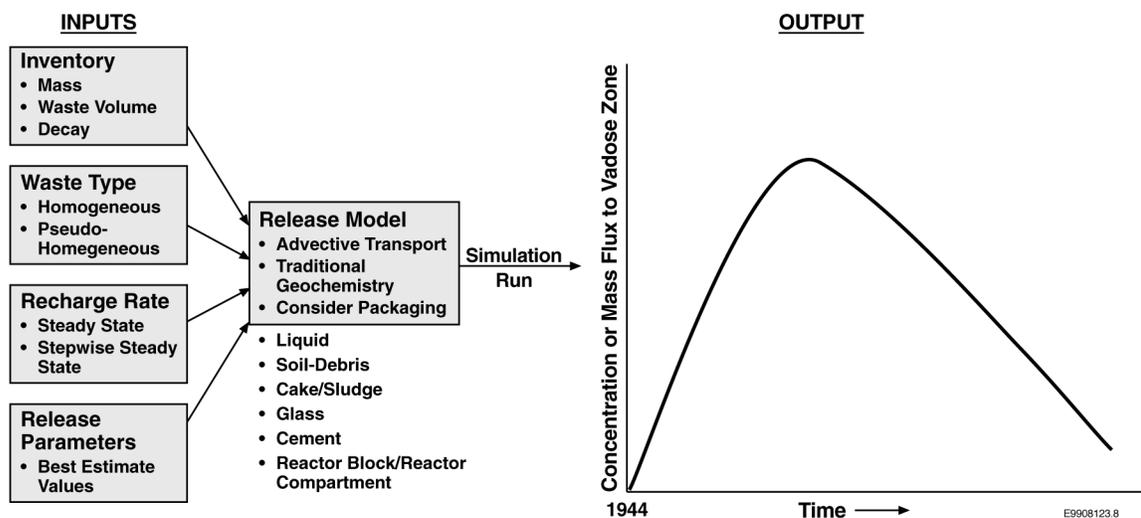
Lastly, recharge can be represented in the release conceptual model as steady state, step-wise steady state, or fully transient. With the steady state, recharge is depicted as an average rate for the length of the simulation. No changes in recharge as a function of time are accounted for in this type of simulation. With the step-wise steady state approach (Kincaid et al. 1998), one can account for losses in waste package containment (e.g., breaching of an engineered barrier in out years that would lead to significantly higher infiltration rate) or long term changes in climate. In a fully transient mode, changes in recharge rate can be depicted over much shorter time intervals. This capability would account for the effects of wet seasons, and would address potential non-linearity in infiltration rates and resulting recharge.

Based on the different options, a collection of release conceptual models is recommended for the SAC (Rev. 0) (as highlighted component of Table B-1). The effects of containment will be ignored, and inventories will be assumed to be homogeneously distributed for different waste sources. An exception is where diffusion is used as the release mechanism (e.g., cement waste form). Depending on the nature of the simulation, recharge will be depicted as either steady or

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step-wise steady state. The overall conceptual model is depicted in Figure B-1. Inputs from the inventory will consist of masses and volumes of waste associated with different waste sources. Simulations will be performed on different waste sources (or combinations of waste sources) using specific release model capabilities (liquid, soil-debris, saltcake/sludge, glass, cement, or reactor block/reactor compartment models). Release parameters (i.e., sorption/desorption, solubility, dissolution, diffusion, leaching, corrosion) will be selected based on the waste source undergoing simulation. Parameter values will represent best estimates for the simulation to be performed. Partition coefficients and solubilities for organics, inorganics, and radionuclides have been reported (DOE 1994, Kincaid et al. 1998). Half-lives for radionuclides of interest have also been summarized (Kincaid et al. 1998). Output from the simulations will be a contaminant concentration or mass flux to the vadose zone.

**Figure B-1. Proposed Release Conceptual Model for the SAC (Rev. 0).**



The proposed conceptual model for the SAC uses existing release models. Selection of an approach consistent with the composite analysis approach allows us to take full advantage of the release model capability developed for that assessment, and lessons learned from the implementation of the capability. Brief descriptions of the release model capabilities to be applied in SAC (Rev. 0) are provided below. Parameters used in the implementation of release models is summarized in Table B-2.

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**Table B-2. Key Parameters for Release Model Simulations.**

MODEL TYPE	MODEL PARAMETER <sup>1</sup>						
	Adsorption/ Desorption	Solubility	Dissolution	Diffusion	Leaching	Corrosion	Decay/ Degradation <sup>2</sup>
Soil-Debris							
Cake/Sludge							
Glass							
Cement							
Reactor Block/ Reactor Compartment							

<sup>1</sup>Hatched block indicates model parameter applies to model type

E9908123.10

<sup>2</sup>Assume no degradation for chemicals

### B.3.1 Liquid Release Model

The liquid release model will be applied to past practice liquid waste sites (e.g., trenches, ditches, ponds, reverse wells, french drains, and cribs) and liquid releases from high-level waste tanks. Contaminants will be assumed to be uniformly distributed in the liquid effluent. Releases will be assumed to occur uniformly over the period of the specific site's operation and discharge of waste to the vadose zone.

### B.3.2 Soil-Debris Release Model

The soil-debris release model will be applied to solid waste burial grounds, including ERDF, and low-level waste burial grounds (government and commercial). The model could also be applied to decommissioned process and storage facilities (Attachment 1). In the model, if contaminant inventories in the source are high enough, leaching of contaminants through the bottom of the source will be controlled by the solubility of the contaminant in the soil water. Otherwise, leaching will be controlled by partitioning of the radionuclides and chemicals between aqueous and sorbed phases. Corrosion as a release mechanism can also be considered for those sites (e.g., burial grounds) that contain large quantities of activated metal.

### B.3.3 Saltcake/Sludge Release Model

The saltcake/sludge release model will be applied to solid wastes remaining in high-level waste tanks, following retrieval of the majority of the waste for separation and disposal. It is assumed that the wastes will be permeable to water and dissolve over time because of the high solubility nature of a major structural component of the waste (e.g., a nitrate salt). As the solid waste dissolves at a constant rate, controlled by the aqueous solubility of the major structural component, the contaminants associated with dissolved portion of the waste form are assumed to be released into the percolating water congruently at constant rates related to their concentration in the waste form.

## **Appendix B – Release Conceptual Model**

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### **B.3.4 Glass Release Model**

The glass release model will be applied to immobilized low activity waste forms generated from the separation of tank high-level waste into high-level and low-level waste fractions. Vitrified wastes are assumed to release contaminants into the infiltrating water through the dissolution of the glass. The aqueous permeability of the glass will be assumed to be low, such that aqueous transport within the waste form itself is essentially zero. Release will be assumed to occur with time by slow dissolution from the exterior surfaces of the glass. The overall rate of dissolution of the waste form will change with time, because the surface area of the waste form changes as the waste form shrinks. All of the contaminants associated with the dissolved portion of the waste form will undergo congruent release into the surrounding pore water at rates related to their concentration in the waste form and the overall waste form dissolution rate at the given time.

### **B.3.5 Cement Release Model**

The cement release model is applied to special waste forms that are found in selected low-level waste burial grounds. The waste form is assumed to have permeability much lower than the surrounding soil. The pore space connectivity in the cementitious waste form is sufficiently high to allow contaminant mobility within the waste form by diffusion. Percolating water is assumed to surround this waste form, and contaminants inside the waste form are assumed to diffuse to the outer surface and enter the percolating water. After allowing for a certain length of time for cement stability, the cement degrades and dissolution and leaching become the dominant mechanisms influencing release.

### **B.3.6 Reactor Block/Reactor Compartment Release Model**

The reactor block/reactor compartment release model will be applied to graphite reactor cores and associated shielding materials and to submarine reactor compartments. It has been shown that graphite comprising the reactor cores is insoluble, and corrodes insignificantly. Release of radionuclides (i.e., carbon-14) from the graphite core is based on a release rate calculated from experimental leach test data. Release of radionuclides within activated metal to the percolating water is assumed to be released by the corrosion process. Release of lead to infiltrating water is assumed to be solubility controlled. Release of constituents to the infiltrating water will be at a constant concentrations until the applied inventories are depleted.

For submarine reactor compartments, the release of radionuclides and chemicals will follow the approach described for activated metals and lead in the reactor block release. Release of radionuclides from activated metals will use a corrosion rate consistent with a corrosion resistant stainless steel.

## Appendix B – Release Conceptual Model

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### B.4 RELEASE MODEL UNCERTAINTY

Uncertainty in the proposed release conceptual model is present in all four of the technical elements (model type, containment strategy, inventory distribution, recharge representation) used to describe the scope of the SAC (Rev. 0).

Uncertainty can arise in the processes of infiltrating water flow and the interactions of contaminants with the infiltrating and surrounding soils within the waste package. The model for SAC (Rev. 0) will assume advective transport of contaminants out of the waste package. Contaminant arrival times at the waste package boundary will be sooner than predicted if interactions (i.e., sorption) with the surrounding soils are allowed to occur. Assuming unobstructed (as opposed to distributed flow; e.g., differences in subsurface permeability of soil layers) of infiltrating water in the release model will predict sooner arrival of contaminants at the waste package boundary. Values of key parameters used in release model simulations (i.e., sorption/desorption, solubility, dissolution, diffusion, corrosion, leaching) also lead to uncertainty. Partition coefficients of soils can vary significantly from one waste source to another, as summarized in Kincaid et al. (1998). Ranges in partition coefficients have been developed for different waste source types at the Hanford Site, and best estimates are provided. Model simulations tend to use the low end of these ranges in order to be conservative. The geochemistry of the infiltrating water influences the concentration of contaminants in that water (if their release is solubility controlled), thereby generating uncertainty. Releases of radionuclides from activated metals are modeled based on corrosion rates of the metals. More realistically, the dissolution rates of corrosion products (e.g., iron hydroxide) govern the release of the radionuclides into the infiltrating water. In some cases, the primary release mechanism for a particular waste form (such as reactor cores) is difficult to determine. This is the case with reactor graphite cores. Graphite is highly impermeable, and does not corrode. Simulations are based on laboratory leach rate data that may not reflect the dominant mechanism of release of carbon-14 from the waste form. Finally, interaction of contaminants released from the waste form are assumed not to interact with the surrounding soil in the SAC (Rev. 0) release conceptual model. In fact, geochemical reactions do occur and could result in increased or decreased contaminant release rates.

Uncertainty also is present within the waste packages (tanks, vaults, steel containers, etc.). For example, high-level waste tanks are designed to remain intact for a certain number of years (design criteria). However, in the real environment, that design life could vary significantly. Such uncertainty affects the length of time that occurs before infiltrating water contacts a waste material, thereby delaying release of contaminants out of the engineered barrier. Application of engineered barriers to waste sources has a similar effect (i.e., delaying contact of infiltrating water with the waste).

The rate and extent to which contaminants are released from waste is highly uncertain because of the non-homogeneity of waste and a lack of understanding of the spatial distribution of waste within waste sources. In the SAC (Rev. 0) conceptual model, the wastes are considered uncontained and uniformly distributed throughout the packages. Uncertainty in release arises because some wastes are contained and (in some cases) are placed in discreet locations in the

## Appendix B – Release Conceptual Model

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waste package. Quantification and reduction in such uncertainty will require a knowledge of the spatial distribution of the waste inventory.

### B.5 ASSUMPTIONS/TECHNICAL RATIONALE AS BASIS FOR RELEASE CONCEPTUAL MODEL

Specific assumptions associated with the release conceptual model are summarized in Table B-3. A number of the assumptions are the result of the absence of (1) sufficient data to estimate inventories, and knowledge of the spatial distributions of waste within the waste packages; and (2) knowledge of the physical/chemical properties that influence release characteristics from the waste.

**Table B-3. Key Assumptions for the Release Conceptual Model. (3 Pages)**

<b>Assumption</b>	<b>Rationale</b>	<b>Impact</b>
Instantaneous response to changes in recharge rate.	Sites are generally shallow and should respond quickly to changes in recharge relative to the 1,000-year study period.	Changes in recharge at deeper sites will occur gradually over many years. Since decreased recharge results in decreased release from the waste form for each of the release models, when recharge rates decrease the model will underestimate the release for the next few years.
Uniform release of contaminants in liquid releases.	Insufficient data are available to justify distributing the mass of contaminants released in liquid discharges in any specific distribution.	If the majority of mass releases occurred early in the operation of liquid disposals, cumulative mass released at the water table would be underestimated. However, in a few hundred years, it can be expected that the cumulative releases would be approximately equal.
Water content in the soil-debris waste form is constant and equal to estimated pre-Hanford soil moisture content of surrounding soil.	Soil hydraulic properties of soil- debris waste forms are generally unavailable.	In the soil-debris release model, given a specific recharge rate, lowering the soil moisture will result in earlier cumulative releases. Using a low moisture content (estimated from the hydraulic properties of adjacent soil and a steady infiltration rate of 5 mm/yr) will result in earlier cumulative releases, except in cases where a barrier reduces the recharge below 5 mm/yr.
Only a single release model is considered for each site.	Inadequate data is available to estimate inventories that may have been disposed in different waste types at the same site. However, tanks were treated as three separate sites: tank leaks; tank losses; and tank residuals.	Highly mobile wastes may be handled separately from less mobile wastes. For instance, highly mobile wastes may be packaged differently (e.g., cement waste forms) and disposed in a solid waste burial ground with less-mobile wastes. A release model will be used that results in the earliest cumulative release.

## Appendix B – Release Conceptual Model

**Table B-3. Key Assumptions for the Release Conceptual Model. (3 Pages)**

Assumption	Rationale	Impact
Soil-debris release models assumed the waste form is continuously mixed.	The parameters and distributions of inventories within the waste forms were highly uncertain. Using a completely stirred tank reactor model is a reasonable approximation.	<ul style="list-style-type: none"> <li>• Completely mixing the waste form can result in earlier releases by sufficiently diluting the inventory to prevent any local controls on the release (e.g., solubility controls around a hot spot in the waste form or organic phases that are immiscible with an aqueous phase). Mixing the waste minimizes competition of contaminants for sorption sites and chemical interaction between contaminants.</li> <li>• Completely mixing the waste form will decrease the early cumulative releases by continuously redistributing the mass into the upper portions of the waste form. Therefore, this is not a conservative assumption. The magnitude of the impact varies for each site. It is most likely to affect releases of highly mobile wastes by delaying their release.</li> </ul>
Leaching occurs by advective transport of the aqueous phase contaminant out of the bottom face of the source zone.	Loss in retardation of contaminant transport due to advective transport is small compared to contaminant retardation that occurs in the vadose zone and groundwater.	Assuming no interaction of the contaminants with soils once they are released from the waste form, contaminants will be released from the waste package more rapidly than if interaction were allowed to occur. The distance that a contaminant travels to breach the waste package is small compared to distance traveled in the vadose and groundwater. The assumption is conservative in that arrival time in the vadose zone and other points along the subsurface environmental pathway will be slightly increased.
The soil debris model assumes the properties of the surrounding soil rather than the waste dictate contaminant release.	The availability of physical and chemical data regarding high surface to volume wastes is limited.	Unconsolidated wastes in the soil-debris waste form type can be sub-divided into those having either high or low surface-area-to-volume (S/V) ratios. Contaminants in the high S/V category (e.g., sludge, soil, spent filters/adsorbents) can have high surface adsorption coefficients. Release from these wastes can be controlled by the waste material itself. The assumption is understood to be conservative because the sorptive properties of the surrounding soil would be lower than those of the waste material.
No credit taken for waste containment in release models.	Stability of containment materials in the natural environment may vary significantly from design criteria. Degradation data are not available for some containment materials.	Materials comprising containment structures (e.g., concrete, wood, steel, iron) vary significantly from waste source to waste source. Unbreached, these materials delay the interaction of infiltrating water with the waste form and, therefore, the release of contaminants into the percolating water. The assumption is thus conservative in its effect on contaminant release from waste sources.

## Appendix B – Release Conceptual Model

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**Table B-3. Key Assumptions for the Release Conceptual Model. (3 Pages)**

<b>Assumption</b>	<b>Rationale</b>	<b>Impact</b>
Best estimate values used for geochemistry parameters in release models.	Geochemical parameters such as $K_d$ can vary significantly among and within waste source terms. Best estimate values are used to reduce the number of model simulations required.	In general, the lower part of the best estimate range is used in model simulations. This approach is considered conservative with respect to simulation results. Conservatism also is practiced for other parameters. For example, a steel with the greatest corrosion rate would be used to simulate release of radionuclides from activated metal associated with a reactor compartment.

### B.6 OUTSTANDING ISSUES

Improvements in the predictive capabilities of release models are dependent on the resolution of several issues. Best results in predictive capability are derived from being able to simulate the release of contaminants from individual source terms, using complex modeling capabilities, and summing the results of those individual release scenarios in ways that address different environmental decision making needs. Absence of complex capability, cost, and simulation run time make such types of application prohibitive. To address this issue, waste sources are composited, reducing the number of simulations that need to be run to more manageable levels. Compositing requires accurate knowledge of the contaminant inventories for each site, so that proper waste source groupings can be established. Kincaid et al. (1998) used compositing techniques for tank wastes, wastes in solid waste burial grounds, and high volume liquid release sites in the 200 Areas to perform analyses and predict release of contaminants to the Columbia River. Most recently, logical waste site groupings were identified for the 200 East and 200 West Areas to meet environmental restoration objectives on the 200 Area Plateau (DOE 1997b, 1999). Rational development of waste site groupings for the 100, 200, and 300 Areas will be needed to ensure appropriate inventory inputs to release model simulations for the SAC (Rev. 0) and beyond.

Increased release model capability may be needed for future revisions of the SAC (Rev. 0). Increased capability includes movement from simple to more complex models, which incorporate multidimensional flow capabilities and account for the interaction of released contaminants with the surrounding waste package matrix (e.g., soil). Similar increases in capability would be needed to address the effects of waste packages on contaminant release and variation in infiltration rates that are most likely to be observed in the near surface environment of the waste packages. Additional research will be necessary to improve the parameterization and data requirements for implementing such capabilities in future versions of the SAC.

### B.7 PROPOSED PATH FORWARD

Based on the different options, a release conceptual model for SAC (Rev. 0) has been selected (see the highlighted component of Table B-1) to consist of the implementation of simple release

## Appendix B – Release Conceptual Model

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models for all simulations in the SAC (Rev. 0). The models will assume advective transport of contaminants out of the waste package, and application of traditional parameters (e.g.,  $K_d$ , solubility, dissolution, diffusion, corrosion, leaching) to simulate release. The effects of containment will be ignored in most cases, and inventories will generally be assumed to be homogeneously distributed for different waste sources. An exception regarding containment would be where diffusion is used as the release mechanism (e.g., for inventories contained in cement waste form). A pseudo-homogeneous option may (in some instances) be applied to inventory distribution (e.g., release from a low-level waste burial ground). Depending on the nature of the simulation, recharge will be depicted as either steady or step-wise steady state.

Implementation of this alternative for the SAC (Rev. 0) has a number of advantages. The alternative allows the potential leveraging of available hardware and software capabilities and available parameter data sets used and lessons learned from implementation of previous assessments (Kincaid et al. 1998). The relative simplicity of the models allows many simulations to be run for different release scenarios, thereby facilitating sensitivity analyses. One disadvantage is that a key process that significantly influences release rates may be missed. The identification of more complex options, coupled with knowledge gained from implementation of SAC (Rev. 0), will help define priorities for where one moves next in release conceptual model SAC capability.

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### **ATTACHMENT 1**

#### **TANK HIGH LEVEL WASTE**

There are 177 large underground tanks (149 single-shell and 28 double-shell) located in 18 tank farms that store high-level waste at the Hanford Site. The tanks currently contain a total of 212 million liters (56 million gallons) of liquid, sludge, and saltcake (Figures 1 and 2). The single-shell tanks had a design life of 20 years. By the late 1980's, 67 of the single-shell tanks were known or suspected leakers, where an estimated 3.8 million liters (1 million gallons) of high-level waste had been released to the soil underlying the 200 Areas. To address this concern, the 28 double-shell tanks were built, and most of the free-standing liquid from the single-shell tanks was transferred to them. Newly generated waste is stored in the double-shell tanks. No leaks are known to have occurred from these tanks.

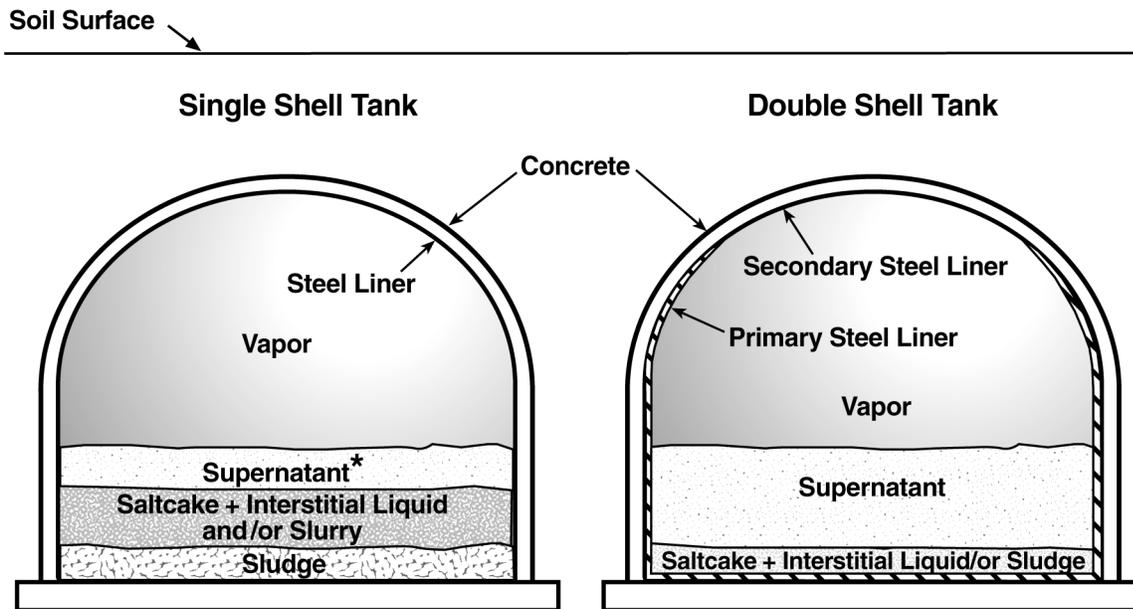
Six alternatives have been assessed for managing these wastes for the long term. At one extreme, no action would be taken. The wastes would be allowed to remain in the tanks under administrative controls. Other actions call for in situ and ex situ treatment of the wastes. A favored alternative is to retrieve the waste and separate it into high- and low-level waste fractions, leaving a minimal amount of waste (as low as 1%) remaining in the tanks. The high-level waste fractions would be shipped offsite to an underground repository, and the low-activity fraction would be vitrified, containerized, and stored on-site in underground concrete vaults (see ILAW below). The remediated tanks would be filled with gravel. The surface of the ground above the tanks would be covered with a multi-layer hanford barrier (DOE 1996c).

#### **IMMOBILIZED LOW ACTIVITY WASTE**

Radioactive wastes stored in 149 single-shell and 28 double-shell tanks at the Hanford Site are to be retrieved and separated into high- and low-level waste fractions. The high-level waste fraction is to be packaged for long-term offsite storage in a high-level waste repository. The ILAW fraction is to be vitrified (i.e., into a silicate glass), placed in containers, and disposed in underground concrete vaults located within (or adjacent to) the 200 East Area. Current planning for the disposal facilities includes a surface cover (to minimize the flow of water or other potential intrusions into the facility) and a sand-gravel capillary barrier to divert water around the waste form (Figure 3). More than 200,000 m<sup>3</sup> (7,000,000 ft<sup>3</sup>) of immobilized low-activity waste from single- and double-shell tanks (equal to a football field covered with packages stacked 76.2 m (250 ft) high is to be disposed in these vaults (Mann et al. 1998).

# Appendix B – Release Conceptual Model – Attachment 1

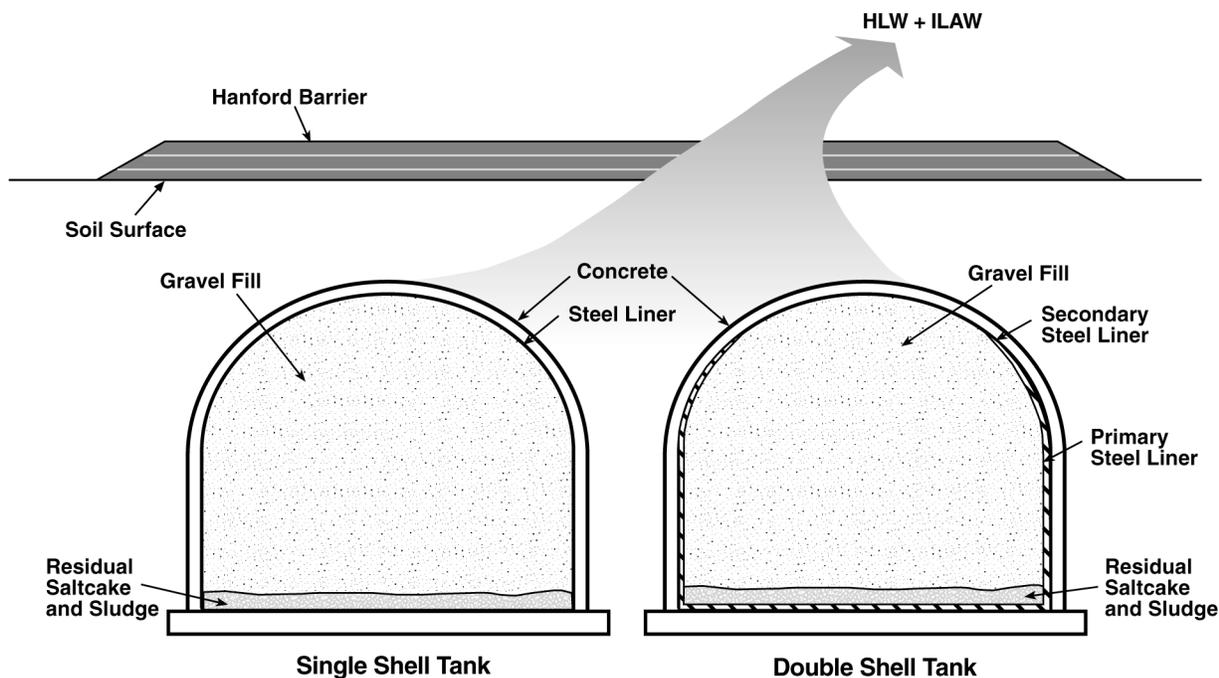
Figure 1. Tank Waste No Action Scenario.



\*Supernatant to be removed in "no action" scenario

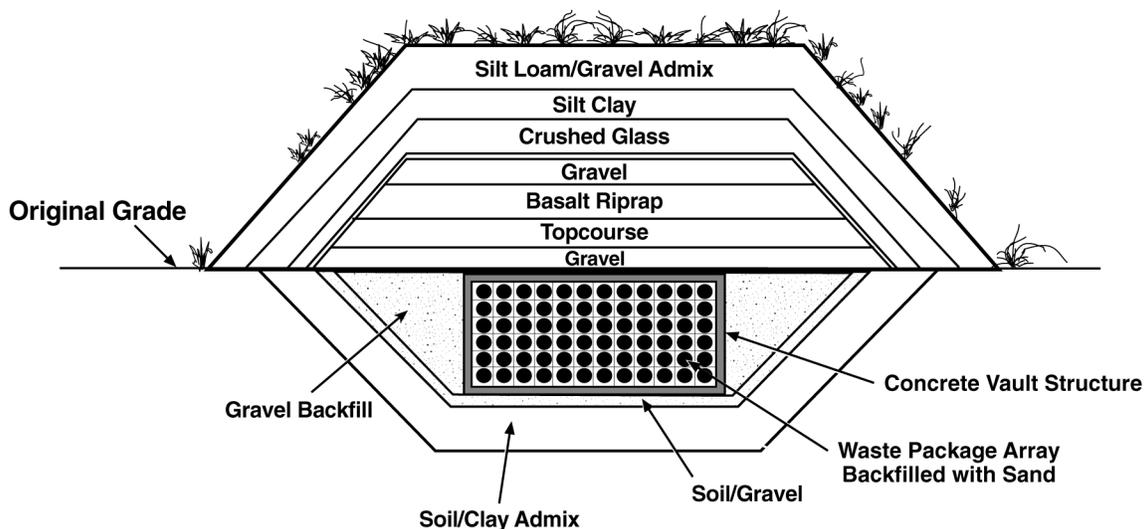
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Figure 2. Tank Waste (Waste Retrieval).



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Figure 3. ILAW Disposal.

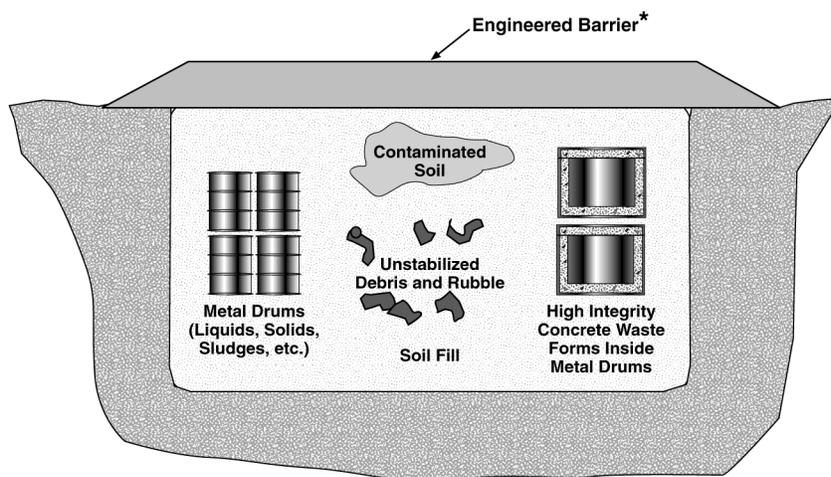


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**SOLID WASTE BURIAL SITES**

Shallow land burial of solid waste has occurred over many years in the Hanford Site's 100, 200, and 300 Areas. These sites (usually unlined) received various wastes from on-site and off-site generators. The physical and chemical characteristics of the wastes include absorbed liquids and solids/sludges, compactable solids (e.g., clothing, paper, plastics, wood), construction materials (e.g., concrete rubble), activated metals, particulates, spent resins, and failed equipment. Adding to the complexity of the waste composition is the potential for the presence of high-integrity cement waste forms (higher inventory waste) (Figure 4) (Wood et al. 1995, 1996).

Figure 4. Solid Waste Burial Grounds.



\*Likely addition to meet future RCRA closure requirements

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## **Appendix B – Release Conceptual Model – Attachment 1**

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### **Low-Level Waste Burial Grounds**

From the early 1940s until 1970, no distinction was made between transuranic (TRU) and low-level waste (LLW). In the early 1980s, low-level liquid organic waste was segregated from LLW and retrievably stored underground. A further categorization of LLW occurred in 1987, when the concept of mixed waste was established. Mixed waste disposal was largely discontinued. Thus, a variety of waste types (i.e., TRU, mixed TRU, LLW, and mixed LLW) have been disposed over the years in low-level waste burial grounds.

This waste is not segregated spatially by type. The waste is usually packaged in carbon steel drums, or wooden boxes. Filled sites are covered with soil to grade and, in some instances, planted with grass to stabilize the soil cover. The trenches are unlined. Retired trenches are likely to receive engineered barriers to meet future RCRA closure requirements. Efforts are being made to consolidate future disposal of this type of waste in the 200 West Area Low-Level Burial Ground (Wood et al. 1995, 1996).

### **Environmental Restoration Disposal Facility**

The ERDF facility is a landfill authorized under CERCLA to receive past-practice waste originating from the Hanford Site's 100 and 300 Areas and, later, from certain units located in the 200 Areas. Waste is transported by truck to the facility and consists of soil and demolition debris contaminated with hazardous, radioactive, and mixed waste. The ERDF consists of individual trenches (cells) that are 21.3 m (70 ft) deep and 4,328 m (1,420 ft) wide and 234.7 m (770 ft) long at the surface. The facility is a RCRA double-lined trench and, at closure, will include a modified RCRA compliant cover (DOE 1994).

### **Processing and Storage Facilities**

Chemical separations plants located in the 200 West and 200 East Areas (i.e., B Plant, T Plant, the 221-U Facility, PUREX, and REDOX) were built to extract plutonium from fuel rods irradiated in the Hanford Site production reactors. Known also as canyon buildings (i.e., B Plant, T Plant, and the 221-U Facility), because of their monolithic size and canyon-like appearance of their interiors, these plants were equipped to use the bismuth phosphate separation process. The 221-U Plant was never used to process plutonium. The PUREX and REDOX facilities used more efficient solvent extraction processes to recover plutonium. The K Basin facility stores spent nuclear fuel from past operation of the N Reactor. The decision in 1992 to deactivate the PUREX plant left approximately 2,100 metric tons of spent fuel in the basins with no means of near-term removal and processing. About 80% of the DOE's spent nuclear fuel inventory is located at the Hanford K Basins. The basins have a history of leaks to the environment, including some greater than a million gallons. The future disposition of these facilities are described below. Greater detail on disposition can be found in Miller et al. (1997), DOE (1997a, 1997b), and Williams (1999).

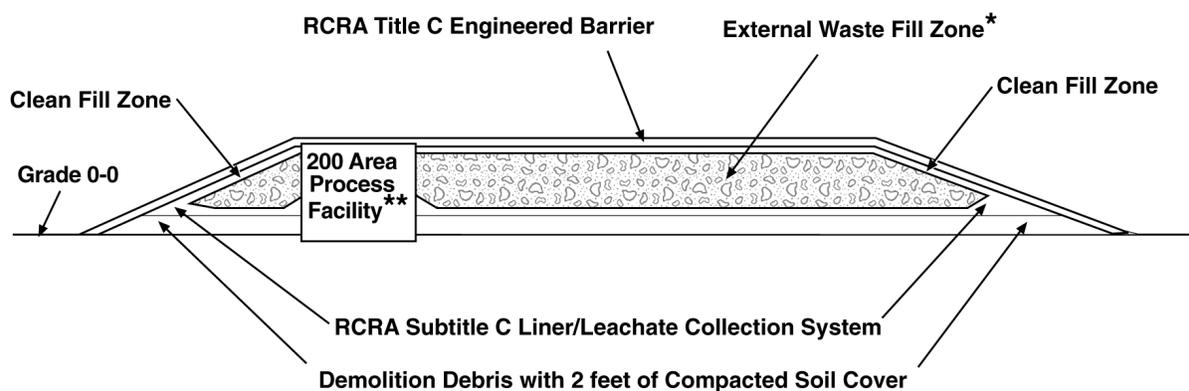
**Disposition Strategy (Processing Facilities).** In 1996, an assessment was performed to determine the long-term disposition of the five main processing facilities. The CERCLA regulatory process was determined to be the appropriate decision-making pathway. Disposition

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options considered included removing the processing facilities, leaving all (or part) of the facilities in situ, and identifying alternative beneficial uses for the facilities. It was identified that the technical approach for dispositioning any of the facilities could be bounded by six basic alternatives.

Two of the six alternatives--entombment with internal (inside decommissioned process facility), or entombment with internal/external waste disposal--appear to be the most favored. These alternatives call for disposal of Hanford Site only remediation wastes (i.e., wastes with radioactivity acceptable for near-surface disposal) inside (or inside and outside) of the facilities, followed by an in situ entombment of the waste-filled facility. The remedial wastes proposed for disposal in the facility would most likely come from 100 Area and 300 Area remedial actions. All access openings to each facility would be sealed. Where possible, void spaces would be filled with compacted earthen materials. Where compacted earthen materials are not practical, sand, fluidized concrete, or grout will be used to eliminate the void space. These alternatives maximize the consolidation of Hanford Site cleanup waste, thereby minimizing impacts in the ERDF expansion area, and achieving greater life-cycle cost effectiveness. Figure 5 depicts this disposal alternative.

**Figure 5. Disposal of 200 Area Process Facility.**



- \*Remediation Wastes from 100 and 300 Areas
- \*\*Contains Process Facility Legacy Waste

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**Waste Management Disposition Strategy (K Basins Storage Facility).** Metallic spent nuclear fuel from the present degraded storage conditions in the 105K East and West basins in the 100 K Area (along the banks of the Columbia River) will be moved to safe interim storage in the 200 Area on the Central Plateau, pending permanent storage in a federal repository. Sludge and debris from the basins will be disposed as low level liquid waste or solid waste. K West floor sludge will remain in the basin. The K Basin area and facilities will be turned over to the RL contractor responsible for ER, following deactivation by the SNF project that is scheduled to be completed by 2007.

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**Release of Contaminants from Processing and Storage Facilities.** A Phase 1 Feasibility Study (FS) for disposition of the 221-U Facility has been initiated to generate the regulatory and technical precedence for future disposition of the other four remaining processing facilities. Upon completion of the Phase 1 screening step, a decision will be made as to whether to proceed with the remaining characterization and completion of the RI/FS process to reach a *Record of Decision* (ROD).

The favored alternatives for 200 Area processing facilities seal all (or most) of the waste inside a hardened concrete structure underneath an engineered barrier similar to the ERDF barrier. Information provided by the ERDF RI/FS and performance assessment fate and transport modeling analyses may have application to parts of the analyses for the 221-U Facility disposition alternatives, and the subsequent disposition of the other four facilities.

When the K Basin facility is released to the environmental restoration contractor, the strategy for remediation could be in place stabilization or perhaps transport to the 200 Area (as is planned for reactor blocks), with a similar disposition scenario as described for 200 Area processing facilities.

**Solid Waste Disposal Sites (inside fences of 200 East and 200 West Areas).** The current strategy is to place engineered barriers at sites located within the 200 Area fence line, and proceed with removal actions at sites outside the fence line (i.e., 200 Area buffer zone) (DOE 1996a).

### **Reactor Blocks (Disposition Strategy)**

Nine graphite core reactors were operated at the Hanford Site between 1944 and 1987, for the purposes of production of plutonium and electricity. A subsequent environmental impact statement (DOE 1989) led to a ROD that called for safe interim storage of eight of the reactor blocks in the 100 Areas (< 75 years), with their gradual dismantlement and movement to a waste burial ground in the 200 West Area. A typical reactor block consists of a graphite moderator stack encased in a thermal shield (lead imbedded cast iron blocks), surrounded by a biological shield (steel/masonite or concrete). The entire block rests on a massive concrete base and foundation. During the decommissioning process, the fuel storage basins associated with each reactor may be dismantled, placed in containers, and buried in the 200 West Area burial ground. If left in-situ, they would be filled with soil.

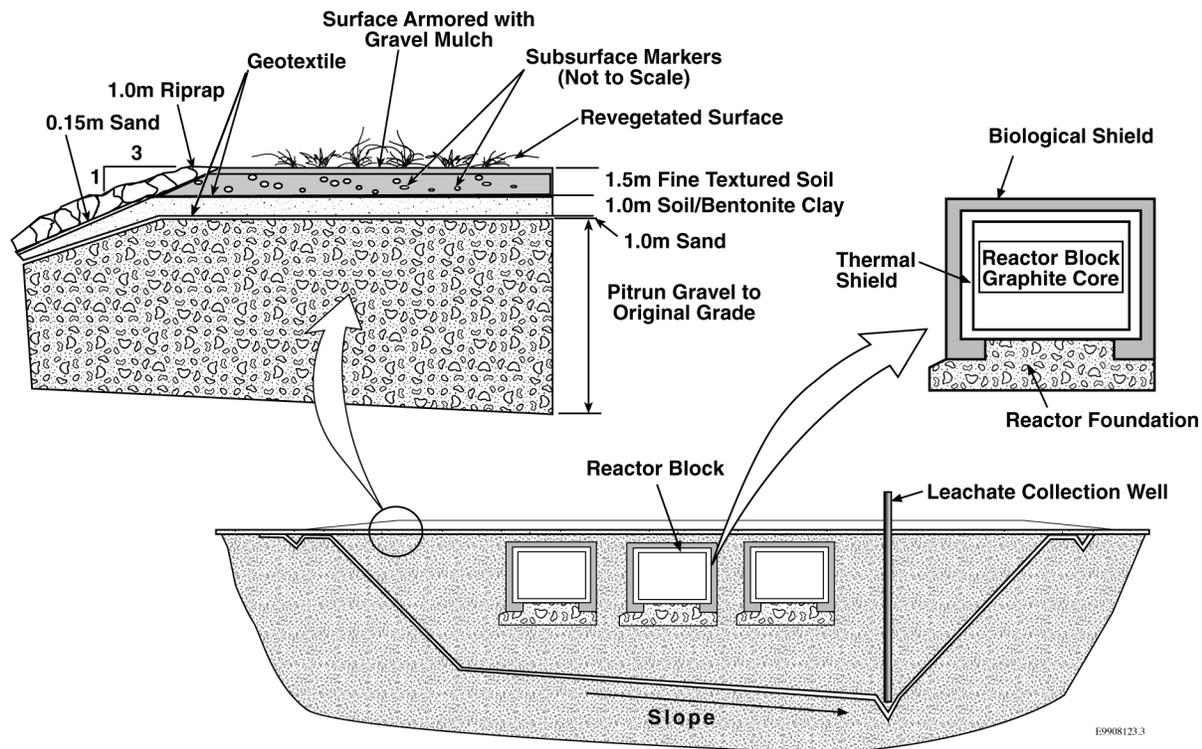
Interim storage of the 105-C Reactor block has recently been completed, and included a roof and siding consisting of aluminum and zinc coatings that provide excellent barrier and galvanic protection. The remainder of the reactors are to achieve this status by 2015. This assumes that the N Reactor will replace the B Reactor in the interim storage process (Day et al. 1997, Palmquist 1998).

For permanent disposal, reactor blocks would be transported to a location in the 200 West Area. Other contaminated materials, equipment, and soils external to the reactor block would be removed, packaged, and transported to the low-level waste disposal site in the 200 West Area. Each disposal site will have a protective barrier, a groundwater monitoring system, and a marker

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system. Disposal sites may be provided with liner/leachate collection systems. The protective barrier is designed to limit infiltration to 0.1 cm/yr. Figure 6 depicts this disposal scenario.

**Figure 6. Reactor Block Disposal.**



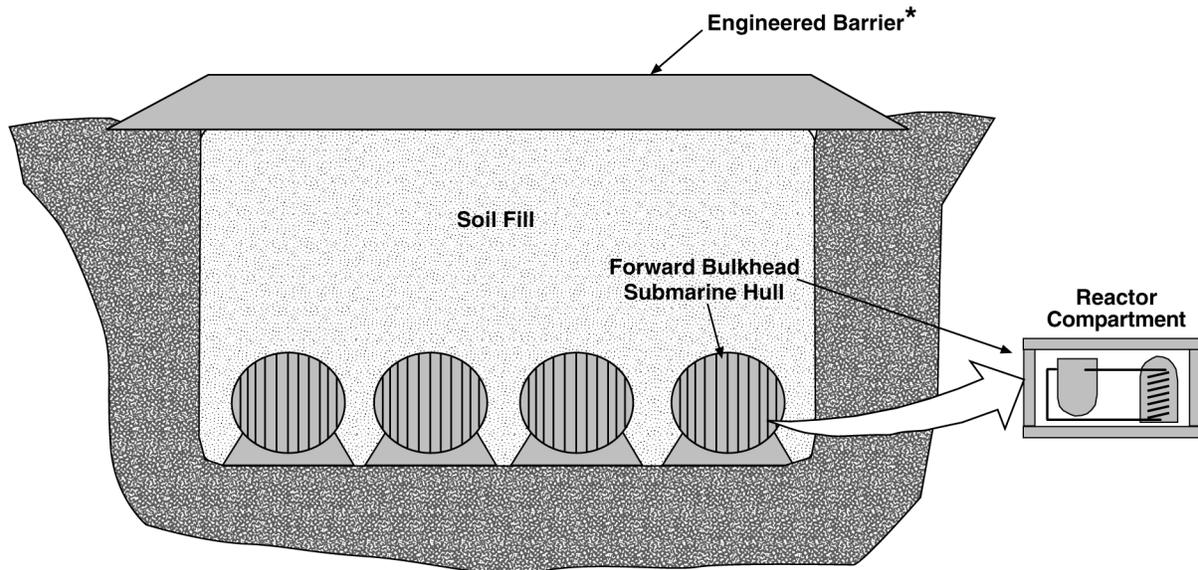
### Disposal of Reactor Compartments

Disposal of defueled nuclear reactor compartments from U.S. Navy nuclear submarine and cruisers occurs in trench 94 of the Low Level Waste Burial Ground (218-E-12B), which is located in the 200 East Area (Figure 7). The initial estimate called for the disposal of 100 compartments, but that estimate has been increased to a proposed waste stream of 220 (DOE 1996b).

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Figure 7. Reactor Compartment Disposal.



\*Likely addition to meet future RCRA closure requirements

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