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3.0 ANALYSIS OF PERFORMANCE

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3.0 ANALYSIS OF PERFORMANCE

3.1 INTRODUCTION

The SST PA provides an assessment of the long-term human health impacts associated with the proposed closure of the WMAs in the 200 Areas of the Hanford Site. As part of that assessment, the postulated events (scenarios) that can lead to adverse human health impacts and the pathways by which contaminants within the final closed system can potentially reach humans in the future must be identified. This chapter provides the methodology developed to assess the scenarios and pathways that were discussed in Chapter 1.0, and describes the approach used to estimate the impacts from the proposed closure action.

Specifically, this chapter describes the models, computer codes, and input data used to analyze the long-term performance of WMAs in the 200 East and 200 West Areas following their closure. For the analyses, the information discussed in Chapter 2.0 was translated into conceptual physical models that incorporate each feature of the natural and engineered barrier systems that impact the system performance. These conceptual models were then translated into numerical models to estimate the risk for each pathway. The best available data were used in the numerical models to estimate the long-term system performance. Where data were not available or were uncertain, assumptions were made and sensitivity cases identified to explore the functionality and capability of each feature of the natural and engineered barrier system.

The strategy for this SST PA was to define and analyze both a reference case and a suite of sensitivity cases. The reference case was developed using the best available information for the physical system and the WMA facilities, and the closure plans for each WMA. Sensitivity cases were defined to explore the relative impact of uncertainties in the models and data, and the assumptions on the estimated health impacts. For example, any potential leaks that may occur during retrieval of tank wastes are not included as part of the reference case, but were considered as part of the sensitivity analyses.

The following topics are discussed in this chapter:

- **Performance Assessment Methodology:** Section 3.2 describes the conceptual models developed for this SST PA, the translation of these conceptual models to numerical models, and the integration of the overall methodology used in the analyses.
- **Numerical Implementation:** Section 3.3 describes the software codes used to calculate the contaminant concentrations associated with different pathways at different locations and the translation of these concentrations as risk estimates to human health.
- **Values and Assumptions:** Section 3.4 describes the values and assumptions used in the numerical calculations to estimate the impacts. This section provides the estimate for the anticipated inventories in the WMAs at closure. This section also describes the values and assumptions associated with a reference case that are developed from the Hanford Site data.
- **Sensitivity Cases:** Section 3.5 describes sensitivity cases that reflect the variability in the system performance or data selected for the reference case.

In an effort to establish credibility and confidence in the data, assumptions, and methods used in the analysis, the following aspects were recognized and addressed:

- Nearly all data, including those for contaminant inventory, geology, hydrology, and geochemistry, were based on site characterization, sampling, measurements, and supplemented by modeling.
- Field-scale processes that are characteristic of highly heterogeneous Hanford Site sediments (e.g., lateral flow and migration) were simulated in vadose zone flow and transport models.
- Independent scientific and technical peer reviews were conducted.
- All computer codes used were benchmarked and verified.
- Sensitivity analyses were conducted to provide insight into the variability and robustness in the estimated impacts to selected assumptions and data choices made with respect to the calculations.

Results using the models and values are presented in Chapter 4.0 for the groundwater pathway scenario, in Chapter 5.0 for intruder scenarios, and in Section 6.5 for the air pathway scenario. Chapter 6.0 also presents the comparison to performance objectives.

An important aspect of the SST PA analysis is the conceptual model for vadose zone flow and transport, and its basis for use in the SST PA. As discussed in Section 3.2.2.4 (vadose zone conceptual model), each heterogeneous geologic unit within the vadose zone is replaced by its homogeneous equivalent. Each geologic unit is assigned its upscaled or effective hydraulic properties. As part of testing of the vadose zone conceptual model, the moisture content data that were collected at the Vadose Zone Test Facility (also known as the Sisson and Lu site) in the 200 East Area were analyzed as part of a separate task. The results of the analyses are presented in “Stochastic analysis of moisture plume dynamics of a field injection experiment” (Ye et al. 2005). A comparison of the observed moisture plume and the simulated moisture plume using an effective unsaturated hydraulic conductivity tensor for the Sisson and Lu site is described in “Estimation of effective unsaturated hydraulic conductivity tensor using spatial moments of observed moisture plume” (Yeh et al. 2005). The upscaled or effective hydraulic conductivities compare well with the laboratory-measured unsaturated hydraulic conductivity data based on small core samples at the site. As discussed in Yeh et al. (2005), the simulated moisture plume does reproduce the general behavior of the observed moisture plume at the field site. Spatial moments of the simulated plume based on the effective hydraulic conductivities are in reasonably good agreement with those for the observed plume (Figure 3 in Yeh et al. 2005), thereby providing an evaluation of the upscaling or effective parameter approach used in the modeling.

3.2 PERFORMANCE ASSESSMENT METHODOLOGY

The SST PA methodology uses conceptual models that are based on the physical system and expected contaminant migration pathways. The conceptual models were then translated into numerical models that are then used to estimate the risk for each pathway.

3.2.1 Overview

Figure 3-1a provides a schematic representation of both the tank system as it will exist at closure and the contaminant migration pathways evaluated in this SST PA. The manmade components of the system that influence contaminant migration include a surface barrier, the tanks and tank infrastructure, the tank fill, and the distribution of waste in the subsurface. The natural components of the system that influence contaminant migration are a number of mostly horizontal stratigraphic layers within the vadose zone and an underlying stratigraphic layer that is part of the unconfined aquifer. Figure 3-1b shows the translation of the conceptual model for the groundwater path to the implementation of the numerical models to calculate the impacts at the WMA fenceline.

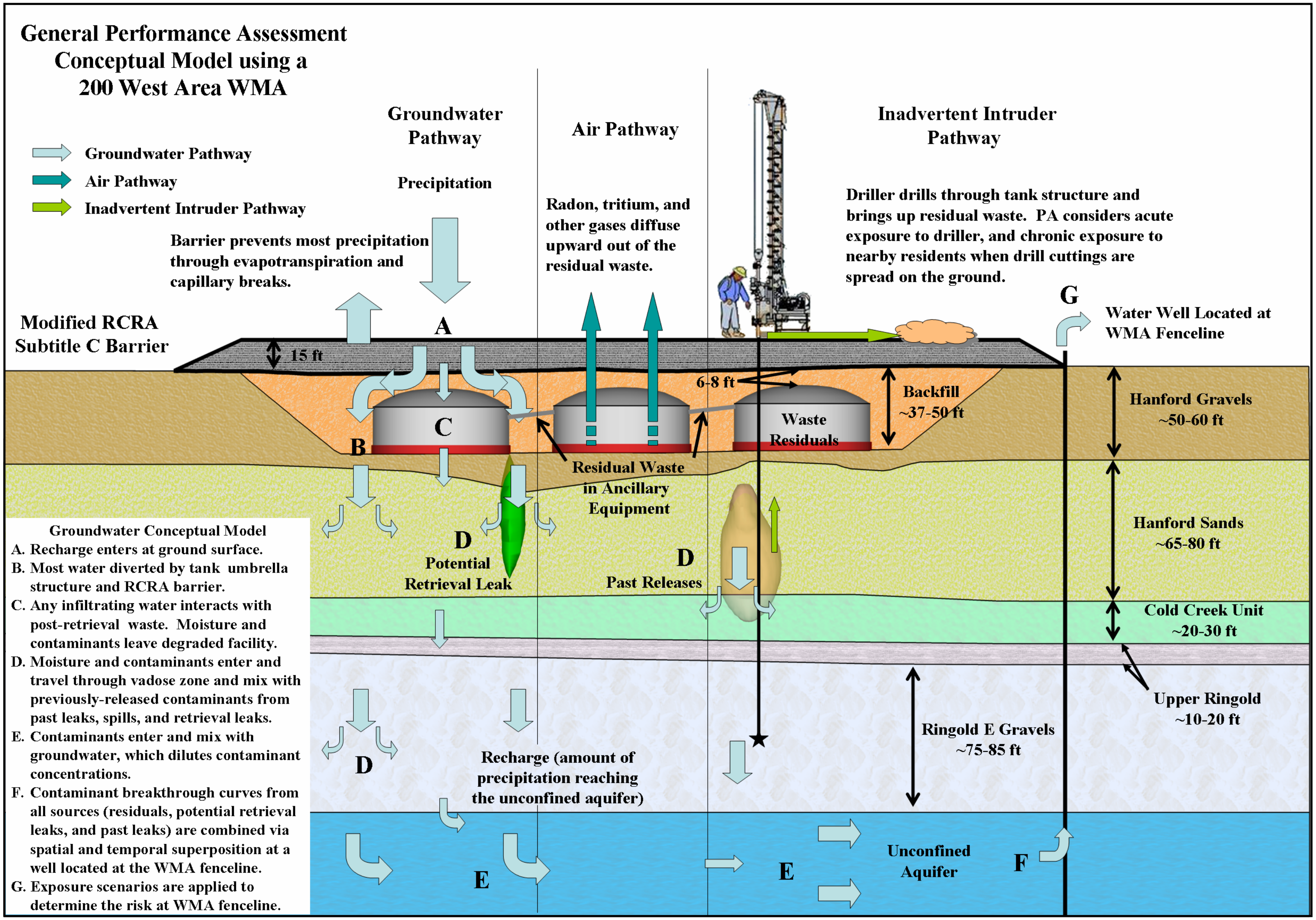
The major pathways for contamination entering the environment are the groundwater pathway, the air pathway, and an intruder pathway. Under the groundwater pathway, it is assumed that water from rain and snowfall enters the subsurface, contacts waste, and carries dissolved contaminants to the unconfined aquifer. Under the air pathway, contaminant gases diffuse from the contaminant sources and into the atmosphere through the Modified RCRA Subtitle C Barrier. Finally, under the intruder pathway, a well is drilled through the contamination located within the tanks or ancillary equipment or in the vicinity of past releases within the vadose zone; the contamination is then brought to the surface where it comes into contact with humans.

Based on the conceptual models for these different pathways, numerical models were developed to estimate the contaminant concentrations within water, air, or soil as a function of time for various scenarios discussed in Chapter 1.0. Functional numerical models cannot be devised to precisely calculate contaminant migration processes in a natural system; simplifying assumptions are required to approximate ubiquitous heterogeneities of the natural system. Also, some aspects of future closure decisions that may affect contaminant migration estimates have not been finalized. Therefore, the numerical modeling approach must be sufficiently flexible to accommodate these uncertainties and to evaluate the effects of different closure decisions on contaminant migration estimates. Finally, contaminant concentration information is used to calculate estimated impacts with respect to the different exposure scenarios discussed in Chapter 1.0.

The SST PA methodology provides deterministic calculations of the estimated impacts from the proposed closure action. The risk impacts are calculated with the numerical models and a set of input values and assumptions that are most representative of the disposal system. This case is referred to as the reference case. The reference case provides the "expected" estimate for how the system may perform given the information available. As more information concerning the waste form, the disposal facility design, and disposal site location is gathered, the definition of the reference case is expected to evolve. The approach used in the reference case is not all inclusive; however, it does provide a reasonable estimate of the expected performance. Selected sensitivity cases have also been used to provide an indication of the sensitivity of the reference case results to assumptions and uncertainty in key parameters.

Figure 3-1. (a) General Performance Assessment Conceptual Model and (b) Numerical Groundwater Conceptual Model

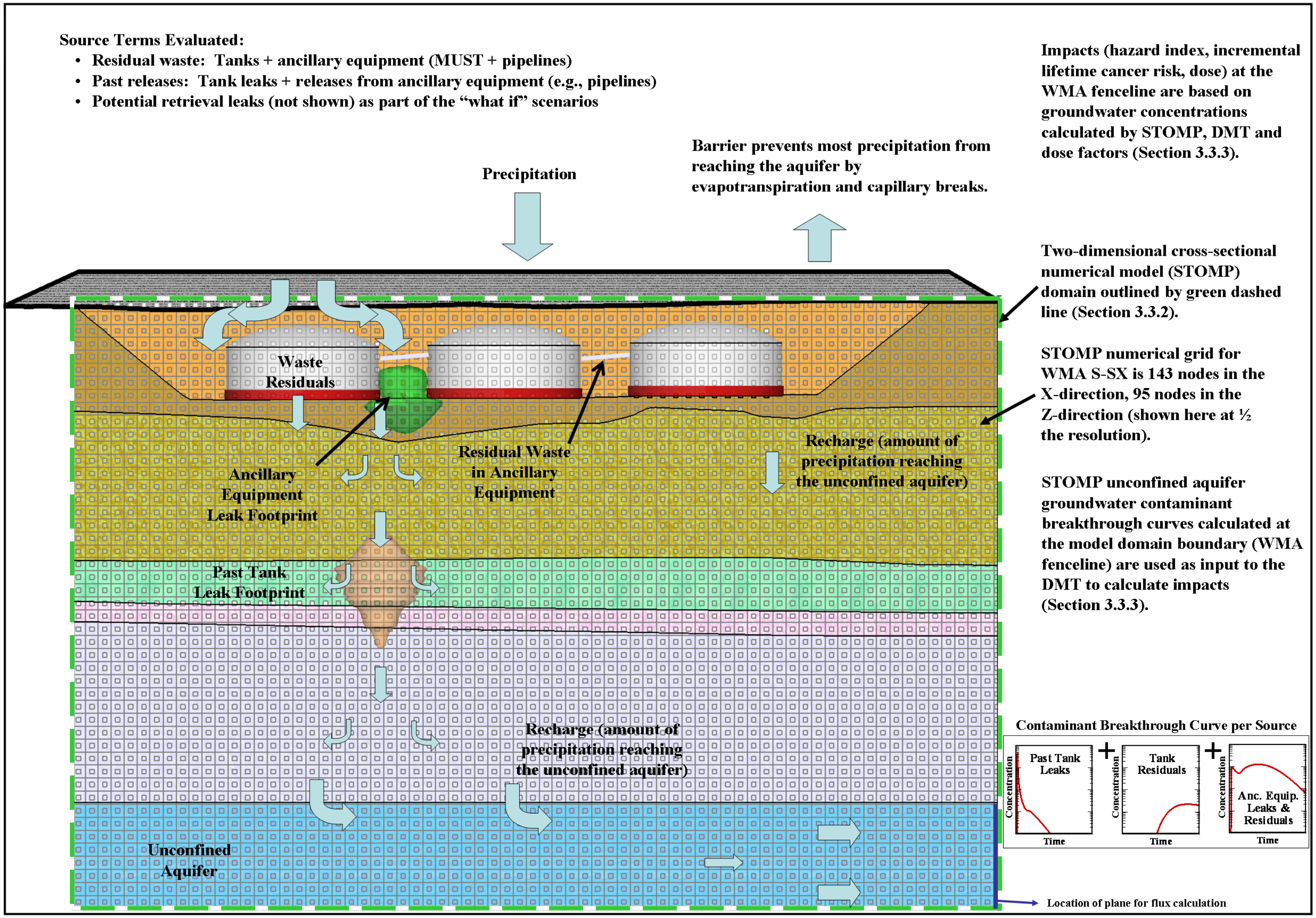
(a)



The geology shown in the figure is specific to the 200 West Area.

Figure 3-1. (a) General Performance Assessment Conceptual Model and (b) Numerical Groundwater Conceptual Model

(b)



The geology shown in the figure is specific to the 200 West Area.

3.2.2 Groundwater Pathway

This section describes the SST PA methodology and the overall modeling approach for estimating the long-term impact and contaminant concentrations in the groundwater. Section 3.2.2.1 presents the overall modeling approach. Section 3.2.2.2 presents details on recharge (infiltration) estimates for various time periods. Section 3.2.2.3 discusses the contaminant release models for various source terms. Section 3.2.2.4 presents an extended discussion on the vadose zone flow and transport model. Section 3.2.2.5 discusses how the groundwater concentration estimates are converted into risk estimates.

3.2.2.1 Overall Modeling Approach

The overall modeling approach for the groundwater pathway is illustrated in Figure 3-2. As part of the closure, an assessment was conducted to evaluate impacts on groundwater resources (the concentration of contaminants in groundwater) and long-term human health risk (associated with groundwater use). The evaluations considered the extent of contamination from the following sources and processes:

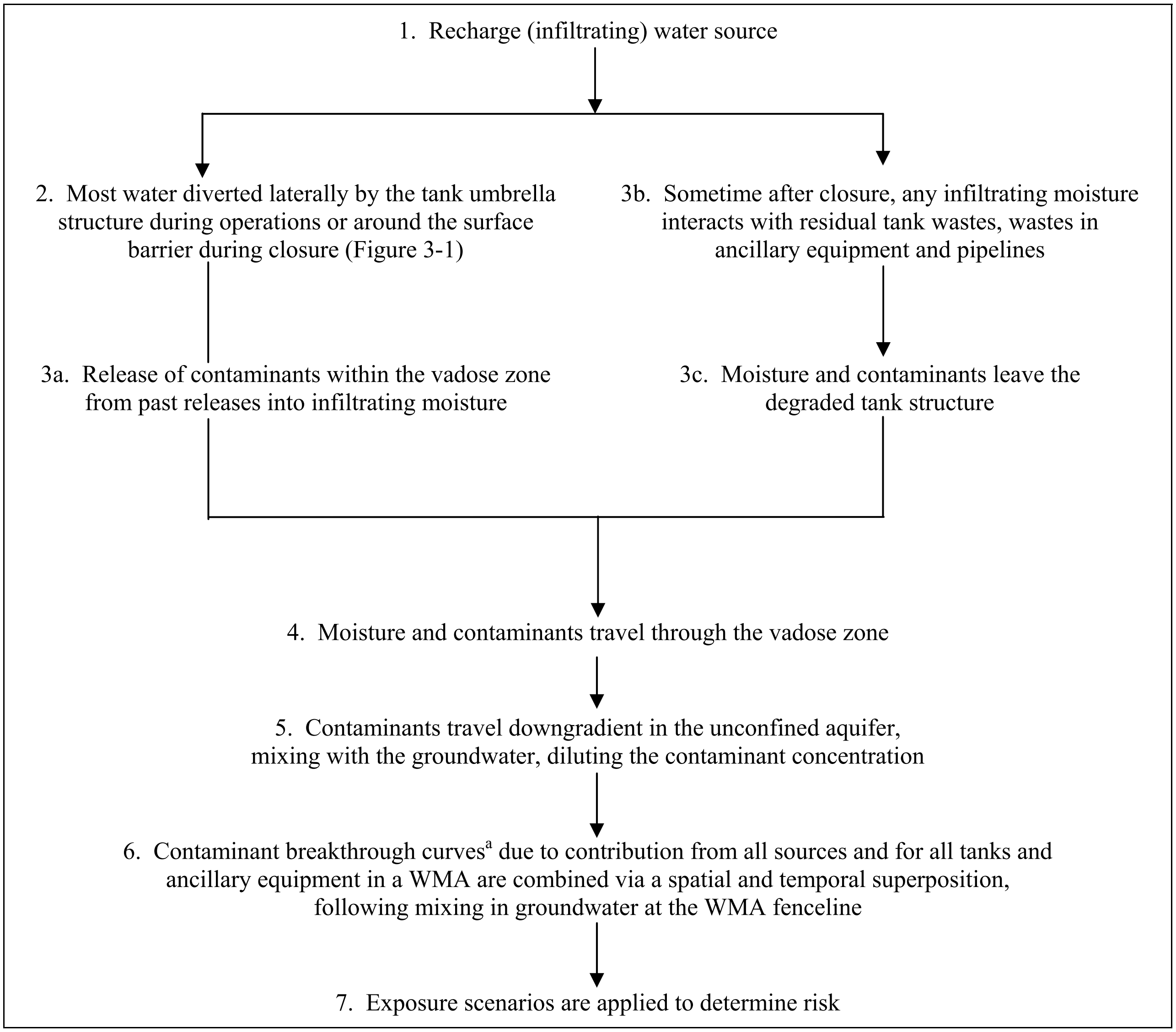
- Residual waste in tanks
- Tank ancillary equipment (i.e., pipelines and MUSTs)
- Past releases (i.e., tank leaks and unplanned releases [UPR] or spills)
- Contaminant movement through the vadose zone to the saturated zone (groundwater)
- Contaminant movement in the groundwater to various calculation points
- Assumed human receptor activities at the WMA fenceline.

> Contaminant sources modeled include:
> - Residual waste in tanks
> - Residual waste in tank ancillary equipment (i.e., pipelines and MUSTs)
> - Past releases (i.e., tank leaks and UPRs) or leaks from ancillary equipment).

As indicated in Figure 3-2, the model assumed that infiltration of moisture from precipitation eventually enters the WMA (step 1), but most of the water is diverted around the tank structure during operations or around the surface barrier during closure (step 2). During the tank farm operational period, contaminants released within the vadose zone from past releases are driven by the infiltrating moisture (step 3a). Following closure, contaminants are released into the vadose zone from the degraded tank structure and ancillary equipment by contact with recharge water (steps 3b and 3c). The infiltrating water, along with contaminants from past releases and residual wastes from steps 2 and 3, travels through the vadose zone (step 4). The contaminants from all sources travel through the vadose zone until they reach the water table and the unconfined aquifer (step 5). The contaminant breakthrough curves (BTC) from residual wastes and past releases are combined via a spatial and temporal superposition (step 6). In the final step of the model, the exposure scenario risk factors are applied to estimated groundwater concentrations to determine risk (step 7).

> Simplifying assumptions were made with respect to contaminant release, recharge, and flow and transport for the vadose zone and unconfined aquifer.

Figure 3-2. Overall Modeling Approach for Performance Assessment for the Groundwater Pathway



[a] *Contaminant breakthrough curves provide the concentration versus time history.*

Key Assumptions. Although much information exists concerning the Hanford Site, much less information exists that is specific to each WMA. The key assumptions were as follows:

- The closure barrier (i.e., Modified RCRA Subtitle C Barrier) for tanks and facilities in a WMA limits infiltration through the waste for a time period that is determined by the time of emplacement of the barrier and the time-dependent barrier performance.
- The fill material in the tanks is cementitious grout. The grout hydraulic properties are not varied during the simulation time.
- The reference case for this assessment assumes that the contaminant release from tank residual wastes is typical of a grouted waste (Section 3.2.2.3.2).

- Calculations are performed for unit curie (Ci) or unit mass (kg) as a source term for each of the three sources (i.e., tank residuals, tank ancillary equipment, and past releases). The contaminant BTC calculations assume proportionality between contaminant source inventory and estimated groundwater contaminant levels.
- Each of the three primary contaminant sources (past releases, tank residuals, and ancillary equipment residuals) can be modeled independently. Release and migration from one source does not alter similar processes occurring with the other sources (see discussion of superposition in Section 3.2.2.4.7).
- For each contaminant source in a WMA, the principle of spatial and temporal superposition is used to obtain a composite contaminant BTC at the WMA fenceline for all sources (Section 3.2.2.4.7).
- Past releases and their contaminant inventories are based on the best available information. In modeling past releases, vadose zone contaminant distributions are used as the initial condition, and the analysis begins in the year 2000.
- Inventories for residual waste in tanks and residual waste in the infrastructure for most cases are the best available estimates at this time.
- The vadose zone is modeled as an aqueous-gas porous media system where flow and transport through the gas phase are neglected (Section 3.2.2.4.7).
- Each heterogeneous geologic unit within the vadose zone is replaced by its homogeneous equivalent (see Figure 3-3 for WMA C and Figure 3-4 for WMA S-SX). Each geologic unit is assigned its upscaled or effective hydraulic properties. A range of K_d values is used to represent sediment-contaminant chemical interaction (Section 3.2.2.4.7).
- Results based on closure risk assessments for WMA C and WMA S-SX are used as the respective templates for analyses for the 200 East and 200 West Area WMAs. Future revisions to this SST PA will have separate analyses for other WMAs.
- Post-closure groundwater flow beneath WMA C was assumed to be parallel to tank row C-103, C-106, C-109 and C-112; similarly, post-closure groundwater flow beneath WMA S-SX was assumed to be parallel to tank row S-101, S-102, and S-103.
- All known contaminants in each WMA were modeled. A number of K_d bins are used to represent the range of sediment-contaminant chemical interaction for the variety of contaminants in various WMAs (Section 3.2.2.4.7).

The timeline for human actions used in this assessment is based on the best estimates available at the time of this writing.

For the groundwater pathway (Figure 3-2), the following models were developed to estimate the risk:

- Numerical models for contaminant release from the contaminant sources associated with the disposal action (step 3 in Figure 3-2) (Section 3.2.2.3)

- Numerical flow and transport model that calculates the flow and contaminant transport through the vadose zone and the unconfined aquifer up to the fenceline (steps 4 through 6 in Figure 3-2) (Section 3.2.2.4)
- Numerical calculation of the estimated risks associated with the public use of the groundwater (step 7 in Figure 3-2) (Section 3.2.2.5).

For this initial PA for the SSTs, detailed conceptual models and corresponding numerical models have been developed for WMA C and WMA S-SX. The results from these numerical calculations have provided estimated contaminant concentrations in the groundwater at the fenceline for each WMA and source term based on a unit curie (Ci) basis. The results are then scaled according to the appropriate inventory estimate. The results from the WMA C calculations are extrapolated to other WMAs in the 200 East Area (WMAs A-AX and B-BX-BY) (Section 3.2.2.4.8). Similarly, the results from the WMA S-SX calculations have been extrapolated to other WMAs in the 200 West Area (WMAs T, TX-TY, and U). A discussion of the justification for such an approach is provided in Section 3.2.2.4.8. Future revisions to this SST PA will include site-specific model calculations, as they are completed, for the contaminant transport to the fenceline for other WMAs.

3.2.2.2 Recharge

The term recharge (infiltration) is used to denote the moisture flux flowing past the evapotranspiration zone (i.e., the plant root zone) that percolates as deep drainage flux to the water table. Recharge is a major driver for contaminant transport from various waste sources to groundwater and to an eventual receptor. Long-term recharge estimates are needed for four different time periods:

- Before construction of tank farms
- During operation of tank farms
- The period during which a fully functional surface barrier is in place
- The period during which the surface barrier is degraded.

Recharge for conditions prior to construction of tank farms is primarily a function of soil type and infiltration characteristics of the native soils. During the operational period, a tank farm ground surface is covered with gravel to prevent growth of vegetation and provide radiation shielding for site workers. Bare gravel surfaces, however, enhance net infiltration of meteoric water, compared to undisturbed naturally vegetated surfaces. Infiltration is further enhanced in tank farms by the effect of percolating water being diverted by the impermeable sloping surface of the tank domes. This umbrella effect is created by the 75-ft (23-m) inside diameter of buried tank domes. Water, shed from the tank domes, then flows down the tank walls into the underlying sediments.

A Modified RCRA Subtitle C Barrier, which significantly reduces the meteoric recharge, is assumed to be in place over the tank farms by year 2032 and to function at its design specification for 500 years. Potential long-term barrier degradation mechanisms include periodic fires that temporarily remove vegetation and transpiration capability. Subsidence or animal burrowing (i.e., biointrusion) can also potentially create localized regions of enhanced moisture via infiltration. Critical components of the near-surface engineered systems include:

1) the vegetative cover to remove water by evapotranspiration, 2) the storage capacity of the surficial sediments to hold water in the shallow zone where it can be readily evapotranspired, and 3) biointrusion barriers to limit human, animal, and plant intrusion into the waste.

3.2.2.2.1 Simplifying Assumptions and Justifications. Recharge potential is enhanced for episodic events during the winter months when the precipitation is at its maximum and the evapotranspiration potential is at its minimum. Vadose zone flow and transport numerical modeling assumes that, for the long-term simulations over tens of thousands of years, the infiltration rates can be averaged on a yearly basis and the discrete nature of the precipitation events can be ignored. The effect of episodic precipitation events on vadose zone flow was investigated as part of a separate task. The results of simulation for a 20-year period of temporally varying precipitation for a surface barrier and a clean graveled surface are included in *Simulations of Infiltration of Meteoric Water and Contaminant Plume Movement in the Vadose Zone at Single-Shell Tank 241-T-106 at the Hanford Site* (Smoot et al. 1989, pp. 18-21). The results show that the temporal variation in drainage can effectively be ignored and an average value can be used with little loss of accuracy. Infiltration with depth through the thick, heterogeneous vadose zone in the 200 Areas dampens the effect of discrete events; therefore, episodic precipitation events can be replaced by an average annual recharge rate. Any potential unfavorable impacts from above-average, short-term infiltration events are not sustained over an extended depth within the thick, heterogeneous vadose zone that is characteristic of the 200 Areas.

Loss of vegetation through fire is temporary (i.e., 1 to 2 years) (Fayer and Szecsody 2004). Also, any potential subsidence is expected to be minimal because of the nature of the underlying material (grouted tanks). Burrowing animals do not create large-volume flow paths under unsaturated conditions (Fayer and Szecsody 2004).

The details of the surface barrier are not explicitly modeled in the numerical model. Instead, an average recharge rate is assumed at the bottom of the Modified RCRA Subtitle C Barrier shown in the conceptual model in Figure 3-1. Average recharge rates (pre- and post-barrier) that are input parameters to flow and transport models are described in Section 3.4.2.

3.2.2.3 Contaminant Release Model

The distribution of residual waste contaminants within the SSTs and ancillary equipment is not known. Contaminants within the tank farm pipelines (residual waste) are assumed to be readily available for transport with the infiltrating water. Residual wastes within the tanks are assumed to be surrounded by grout during the closure process. Release of residual wastes from MUSTs is modeled similarly to release of tank residual wastes.

Upon closure of a WMA, contaminants will be located either in the soils surrounding or beneath the tank farm structures, or within these structures. The contaminants currently residing within the vadose zone soils are from past releases (i.e., tank leaks and UPRs) during tank farm operations. Wastes currently residing within the vadose zone are therefore distributed over varying dimensions and depths. Two types of contaminant releases are considered: 1) instantaneous release (e.g., from past releases) and 2) releases occurring over an extended period (e.g., from residual waste). In the first case, the entire inventory is available for contaminant transport immediately. In the second, the contaminants are available for transport only slowly, and the complete inventory may be released over thousands of years.

First, Section 3.2.2.3.1 describes modeling for release of contaminants from past releases. Section 3.2.2.3.2 presents a similar discussion for release of contaminants from residual tank waste. Section 3.2.2.3.3 describes modeling for release of contaminants from ancillary equipment. Simplifying assumptions made, as well as their justification for use in modeling contaminant release, are presented within each of the three following sections.

3.2.2.3.1 Past Releases. For past release contaminants within the vadose zone, release rates are dependent on contaminant-specific sorption and solubility reactions. In some locations, the dominant reactions affecting contaminant release have changed over time primarily because tank waste chemistry differed from that of ambient soil water and temporarily overwhelmed ambient equilibrium soil water conditions. These chemical perturbations were most significant at the time of the leak event and shortly thereafter, and at locations closest to the leak origin. However, the chemical buffering capacity of soil eventually eliminates tank chemistry influence, and releases are controlled by the ambient geochemical environment.

Each of the three primary contaminant sources (i.e., past releases, tank residuals, and ancillary equipment residuals) can be modeled independently. Release and migration from one source does not alter similar processes occurring with the other sources (see discussion of superposition in Section 3.2.2.4.7).

Chemical reactions that may retard contaminant release are assumed to be those controlled for the most part by the ambient environment. Even though enhanced mobility for several constituents has obviously occurred (e.g., cesium at tank SX-108 [Knepp 2002a], europium in numerous locations such as near tanks T-106 and TX-107 [Myers 2005]), field data suggest that contaminants within the deeper vadose zone behave chemically according to prevailing ambient conditions. Specifically, desorption experiments and solids characterization data from recent characterization borehole sediments (e.g., Knepp 2002a) show that cesium-137, strontium-90, and uranium are now largely immobile in vadose zone soil (i.e., consistent with the ambient geochemical environment).

The past releases including tank leaks and UPRs are listed in Corbin et al. (2005). The list is assumed to capture all major contaminant releases to the soil. Site characterization data are available (Knepp 2002a, 2002b) for the distribution and depth for the contaminant plumes in the vadose zone for various past releases.

Simplifying Assumptions and Justifications for Past Releases. For past releases, the following simplifying assumptions were made:

- One homogeneous contaminant distribution over one waste volume size and depth interval (based on field data from recently drilled boreholes) is assumed for all past releases within each WMA.
- The entire leaked inventory is readily available for transport with the infiltrating water where transport is only limited by the chemical adsorption to the soils.

Actual waste distribution can only be approximated because of limited field data. The use of one contaminant distribution to represent all tank leaks within a given WMA is justified based on the results of previous analyses (e.g., Knepp 2002a, 2002b). These analyses show that an assumed distribution has little effect on mass flux estimates because all contaminant inventories are contacted and migrate to groundwater at about the same rate. The depth locations for the contaminant distribution were selected to be representative of currently measured depths in the 200 East and 200 West Areas. Depending on K_d and timing of barrier placement, the depth location of the past tank leak within the vadose zone, for a given recharge rate, may impact the timing for the contaminant to reach the unconfined aquifer.

> One homogeneous contaminant distribution over a single waste volume size and depth interval (based on field data from recently drilled boreholes) is assumed for all past releases within each WMA. The entire leaked inventory is readily available for transport with the infiltrating water.

The explicit methodology for estimating the contaminant release from each past release is based on the contaminant distributions and depths discussed in Section 3.4.3.1.1, and the use of the WMA C and WMA S-SX results as templates for the remaining WMAs in the 200 East and 200 West Areas, respectively (Section 3.2.2.4.8). More site-specific distributions will be used in later revisions of the SST PA.

3.2.2.3.2 Release of Contaminants from Residual Tank Wastes. The final tank residual waste configuration and inventories will be dependent on actual retrieval practices that remain to be applied to the tank waste. However, as discussed in Chapter 1.0 regarding defense in depth as applied to tank farm closure, the engineered barrier consists of a surface barrier and the grouted tank structure.

Simplifying Assumptions and Justifications for Residual Tank Wastes. The following simplifying assumptions were made for the contaminant release of residual tank wastes:

- Contaminant release from residual wastes in the tanks is dominated by diffusional processes.
- An analytical model for diffusion was used for residual waste contaminants released from tank bottom as a planar source (neglecting tank structure details and any future cracking that may occur within the grouted tank).
- Contaminant specific sorption and solubility were not modeled.
- The source location for the contaminant release from tanks was assumed to be directly beneath the tank.

Tank residual waste is largely encapsulated by low permeability materials and, therefore, is unlikely to be exposed to large amounts of recharge water for significant times beyond the closure date. Therefore, diffusion is assumed to be the dominant mechanism for contaminants to be released from the tanks into the surrounding soils. A simple analytical model (described below) has been selected to represent the contaminant release of residual waste from the closed tank system. Tank waste residuals reside in a grouted block consisting of the tank shell and grouted interior. A mixing zone with a corresponding mixing length of contaminated grout was assumed for purposes of calculating diffusional flux from a contaminated zone. Diffusional flux at the grout soil barrier is a boundary condition in the vadose zone flow and transport model. Diffusion through the steel liner was not explicitly analyzed; cracks in the grout, steel liner, and outer concrete shell were not simulated. These factors were not simulated because too little characterization data are available to simulate the actual conditions. The simulation results were bounded by a sensitivity case, i.e., a release model where the grout was assumed to have the hydraulic properties of backfill material (i.e., sand and gravel) and the tank residuals were assumed to be readily dissolved in the infiltrating moisture (Section 3.5.4.3).

> Contaminant release from residual wastes in the tanks is dominated by diffusional processes. An analytical model for diffusion was used for residual waste contaminants released from tank bottom as a planar source. This assumption neglects tank structure details and any impacts from future cracking that may occur within the grouted tank.

No chemical reactions between contaminants and grout were explicitly modeled in the analysis. For several contaminants (e.g., uranium), chemical reactions that reduce release rates do occur. By ignoring these effects, a release rate that was larger than expected was calculated.

The contaminant release rate from the ancillary equipment pipelines was assumed to be equivalent to the normalized release rates estimated for past tank leaks. A better estimate for the actual release rate must await additional characterization of the waste quantities and their locations within the ancillary equipment, and a better understanding of the planned final closure conditions. For this initial SST PA, on the basis of available information, the inventory associated with all pipeline waste was assumed to be located at a depth of 9 m (30 ft) bgs and uniformly distributed over a horizontal width of 7 m (25 ft). For MUSTs, the release of residual wastes was assumed to be equivalent to the contaminant release of tank residual wastes (Section 3.2.2.3.3).

As stated earlier, a diffusion-dominated release model was used to simulate the release of contaminants from stabilized (e.g., grouted tank) wastes for the reference case. In the absence of little or no advection through the tank waste, the release can be modeled as a diffusion-limited process. The diffusion from cylindrical containers leads to an expression for flux that contains infinite series (Crank 1975; Kozak et al. 1990). The analytical solution used is for one-dimensional diffusion through the tank bottom for a semi-infinite medium with the concentration C_0 throughout, initially, and with zero surface concentration, as follows:

$$C = C_0 \, erf \frac{x}{2\sqrt{(D_e t)}} \qquad \textbf{Eq. 3.1}$$

where:

C = estimated concentration
C_0 = initial concentration
x = distance
erf = standard error function
D_e = effective diffusion coefficient of the contaminants in the waste form
t = time.

The rate of loss of diffusing substance per unit area from the semi-infinite medium when the surface concentration is zero is given by Equation 3.2:

$$\left(D_e \frac{\partial C}{\partial x}\right)_{x=0} = C_0 \sqrt{\frac{D_e}{\pi t}} \qquad \textbf{Eq. 3.2}$$

Equation 3.2 has the form of diffusional mass transfer based on leaching theory. The simplified release model leads to the following form:

$$q = A\, C_0 \sqrt{\frac{D_e}{\pi t}} \qquad \textbf{Eq. 3.3}$$

where:
q = release rate from a single waste cell (Ci/yr)
A = effective surface area of a single cell
C_0 = concentration in a cell.

The residual waste is likely contained in various cells with differing sizes and shapes. For the release model used herein, the cells were assumed to be of the same size and shape so that the diffusive release rate, Q, from all residual wastes in a tank can be based on Equation 3.4:

$$Q = C_0 \sqrt{\frac{D_e}{\pi t}} \sum_{i=1}^{n} A_i = C_0 A_t \sqrt{\frac{D_e}{\pi t}} \qquad \textbf{Eq. 3.4}$$

where:
n = the number of cells
A_i = the surface area of individual cells
A_t = the total surface area.

Assuming that the cells are of constant size:

$$I = C_0 \sum_{i=1}^{n} V_i = C_0 V_t \qquad \textbf{Eq. 3.5}$$

where:
I = the total inventory
V_i = the volume of i-th cell
V_t = the total volume of all cells.

Combining the preceding equations:

$$Q = I \frac{A_t}{V_t} \sqrt{\frac{D_e}{\pi t}} \qquad \textbf{Eq. 3.6}$$

Recall that the real system consists of grout-filled tanks that have residual tank waste located predominantly on surfaces within these structures. Equation 3.6 is a reasonable approximation as long as the diffusional release time is much greater than the vadose zone travel time. See Section 3.4.3.1.2 for details on selection of A_t/V_t values.

3.2.2.3.3 Release of Contaminants from Ancillary Equipment. The final tank residual waste configuration and inventories within the ancillary equipment will be dependent on actual retrieval practices that remain to be applied to the ancillary equipment.

Simplifying Assumptions and Justifications. The tank ancillary equipment is broadly separated into pipelines and MUSTs. The following simplifying assumptions were made for the contaminant release of residual wastes from the ancillary equipment:

- The contaminants within the tank farm ancillary equipment (residual waste) pipelines were assumed to be readily available for transport with the infiltrating water.
- Details on the distribution of tank farm pipelines within the WMA are ignored. For pipelines, the location of inventory in the numerical simulations was assumed to be represented by a homogeneous distribution at a depth of approximately 9 m (approximately 30 ft) and extending horizontally for approximately 7 m (approximately 25 ft). The contaminant release from other ancillary equipment (e.g., MUSTs) was assumed to be equivalent to release of tank residual wastes, as discussed in Section 3.2.2.3.2.

Again, the use of one contaminant distribution to represent all pipeline inventory sources within a given WMA is justified based on the results of previous analyses (e.g., Knepp 2002a). These analyses show that an assumed distribution has little effect on mass flux estimates because all contaminant inventories are contacted and migrate to groundwater at about the same rate. The use of a depth of 9 m (30 ft) to represent the inventories for pipelines is justified because nearly all such ancillary equipment is located just beneath the surface. The assumption that contaminants within pipelines are readily available for transport with the infiltrating water is expected to result in higher than expected concentrations in the groundwater. Also, the barrier that the piping currently provides to infiltrating water is neglected in the SST PA analysis.

> The contaminants within the tank farm ancillary equipment (residual waste) pipelines were assumed to be readily available for transport with the infiltrating water. The release of contaminants within the MUSTs (residual waste) is modeled as a diffusional release.

While final closure plans have not been identified, plans for grouting of some ancillary equipment (MUSTs) are being discussed. The use of the residual tank waste model to represent the release of residual tank waste from MUSTs is believed to be representative of the MUST contaminant release. Differences between MUSTs and SSTs include: 1) the location of the available inventory in MUSTs is not as deep as the tank waste residuals inventory, 2) the footprint for each MUST is typically less than a tank footprint resulting in different shadow effects for recharge, and 3) the contaminant release from the MUSTs may have different values for A_t/V_t when compared to the waste tanks (Section 3.2.2.3.2).

3.2.2.4 Vadose Zone Moisture Flow and Contaminant Transport Considerations

As discussed in Chapter 1.0 with respect to defense in depth, the vadose zone beneath a WMA is considered a natural barrier. Once contaminants enter the vadose zone, the low recharge (infiltration rate) controlled by the surface cover, the thickness of the vadose zone between tank bottom and the unconfined aquifer, and the soil-contaminant interaction prevent all but the least reactive contaminants from reaching the unconfined aquifer for thousands of years.

3.2.2.4.1 Overview. This section provides an overview of major features that affect flow and transport within the vadose zone underlying a WMA. The transport of contaminants from their locations within the closed system to the groundwater is a complicated process that depends on data and assumptions made for the following physical systems:

- WMA structures
- Vadose zone beneath a WMA.

First, this section describes the WMA facility structures (Section 3.2.2.4.2) important to the SST PA methodology. This is followed by a description of vadose zone stratigraphy (Section 3.2.2.4.3), hydraulic properties (Section 3.2.2.4.4), and geochemical effects (Section 3.2.2.4.5) that impact contaminant transport. Next, an overview is presented of the vadose zone flow and transport numerical model used in the SST PA (Section 3.2.2.4.6). Finally, a detailed justification is provided of important assumptions and simplifications of the vadose zone flow and transport model (Section 3.2.2.4.7).

3.2.2.4.2 Waste Management Area Structures. Section 2.4 provides a description of the engineered systems and barriers common to the WMA. The physical system includes the closure barrier and the complex structures that make up the closed WMA. These structures include the tank structures and the ancillary equipment that includes piping, diversions boxes, and other systems that support tank farm operations. These complex structures impact not only the release of contaminants but also the flow of moisture through the system. Moisture is one of the major transport mechanisms for moving contaminants from the closed system to the groundwater. Within the shallow subsurface, moisture fluxes are non-uniform because grout-filled tanks divert moisture flow between the tanks and increase flow rates in these regions. The varying moisture fluxes, however, even out within the deep subsurface below the tanks.

For the conceptual model, the following simplifying assumptions were made:

- The impact of the closure barrier on moisture flow was approximated by an assumed recharge rate into the facility (Section 3.2.2.2).
- The impact of the tanks on moisture flow was handled by assuming that the tanks are impermeable structures that divert flow.
- Details associated with all ancillary equipment on moisture flow were neglected.

The justification for using an estimated recharge rate is provided in Section 3.2.2.2. Also, the justification for neglecting the structural impacts on contaminant release is discussed in Section 3.2.2.3.2. The long-term performance of the tank structures as hydraulic barriers to the flow of moisture within the closed system is not known. It can be hypothesized that eventually cracks will form in the concrete and the steel will degrade at some point in the future.

The tanks are modeled as impermeable structures that divert infiltrating water around the structure.

3.2.2.4.3 Vadose Zone Stratigraphy beneath Waste Management Areas. The vadose zone underlying tank farms consists of several heterogeneous layers of sedimentary units. The layers vary in thickness at different locations (Chapter 2.0). Also, the depth to the water table varies with location. The 200 West Area WMAs are distinguished from the 200 East Area WMAs primarily by the presence of a well-developed calcium carbonate-rich caliche layer (Plio-Pleistocene unit) that has been a relatively effective barrier to contaminant transport from past tank leaks in the vertical direction. Also, clastic dikes (anomalous, subvertical linear features composed of layers of differing particle size distributions) occur that extend up to tens of meters in length and can cross cut the major layers. These features are generally less than 1 m wide.

The geologic cross-section used in WMA C modeling is shown in Figure 3-3. The sedimentary sequences overlying the basalt beneath the WMA C are, from top to bottom:

- Backfill (sandy gravel)
- Hanford formation – upper gravelly sequence (H1 unit, gravelly sand)
- Hanford formation – sand sequence (H2 unit, sand)
- Hanford formation – lower gravelly sequence (H3 unit, gravelly sand)
- Undifferentiated Plio-Pleistocene unit gravel (PPlg) and/or Ringold Formation Unit A [PPlg/(R) unit].

The geologic cross-section used in WMA S-SX modeling is shown in Figure 3-4; the numbers noted with each geologic unit in Figure 3-4 are the material type numbers used for modeling purposes. The sedimentary sequences, from top to bottom, are:

- Backfill (sandy gravel)
- Hanford formation – upper gravelly sequence (H1 unit, gravelly sand)
- Hanford formation – sand sequence (H2 unit, sand)
- Plio-Pleistocene unit – silty very fine sand to sandy silt
- Upper Ringold Formation – sand-dominated facies consisting of slightly silty coarse to medium sand
- Ringold Unit E – Ringold Formation comprising the vadose zone and upper part of the unconfined aquifer consists of slightly silty coarse- to medium-grained sandy gravel with intercalated gravelly sand.

3.2.2.4.4 Hydraulic Properties. Even though no site-specific data are available on soil moisture characteristics for tank farm sediments, data catalogs are available for 200 Areas soils. For this work, data on laboratory measurements for moisture retention, particle-size distribution, saturated and unsaturated hydraulic conductivity, and bulk density for individual stratum were based on data for similar soils in the 200 East and 200 West Areas. Details on vadose zone flow and transport properties are provided in various data package reports (*Modeling Data Package for an Initial Assessment of Closure of the C Tank Farm* [Khaleel et al. 2006a] and

Modeling Data Package for an Initial Assessment of Closure of the S and SX Tank Farms [Khaleel et al. 2006b]). For each stratum defined by the stratigraphic cross-sectional model, the small-scale laboratory measurements were upscaled to obtain equivalent horizontal and vertical unsaturated hydraulic conductivities as a function of mean tension (Khaleel et al. 2002). In addition, to reflect field conditions, the laboratory-measured moisture retention data were corrected for the presence of any gravel fraction in the sediment samples (Khaleel and Relyea 1997). As with flow modeling, each stratum was modeled with different transport parameters (i.e., bulk density, diffusivity, and dispersivity). See Section 3.4.4 for additional details on the equations for hydraulic properties and parameters used for the flow and contaminant transport calculations.

3.2.2.4.5 Geochemical Effects. Contaminant migration rates are element-specific because of the varying degrees of their chemical reactivity with soils (Krupka et al. 2004). Some contaminants are largely non-sorbing (i.e., technetium) and migrate with recharge water. Others are highly reactive and migrate very slowly (i.e., cesium).

Chemical reactions that occur when contaminants interact with soil solid phases and retard contaminant migration relative to water flow through the vadose zone are represented by single sorption (K_d) values (Section 3.2.2.4.7). Different K_d values are considered for particular contaminants, but only over a limited range (0 to 5 mL/g). The K_d value is the ratio of contaminant mass attached to soil solids versus mass dissolved in solution. The advantage of this approach is that K_d values can be easily incorporated in modeling transport. The disadvantage is that K_d values are entirely empirical and are used to represent many different kinds of chemical reactions that are dependent on the contaminant of interest, the soil solid phases present in the vadose zone, and the soil water chemistry. The effects of physical variables (moisture content and gravel fraction) and reactions (colloid formation and migration) are also incorporated in the K_d concept.

Figure 3-3. Northwest-Southeast Cross-Section through Waste Management Area C

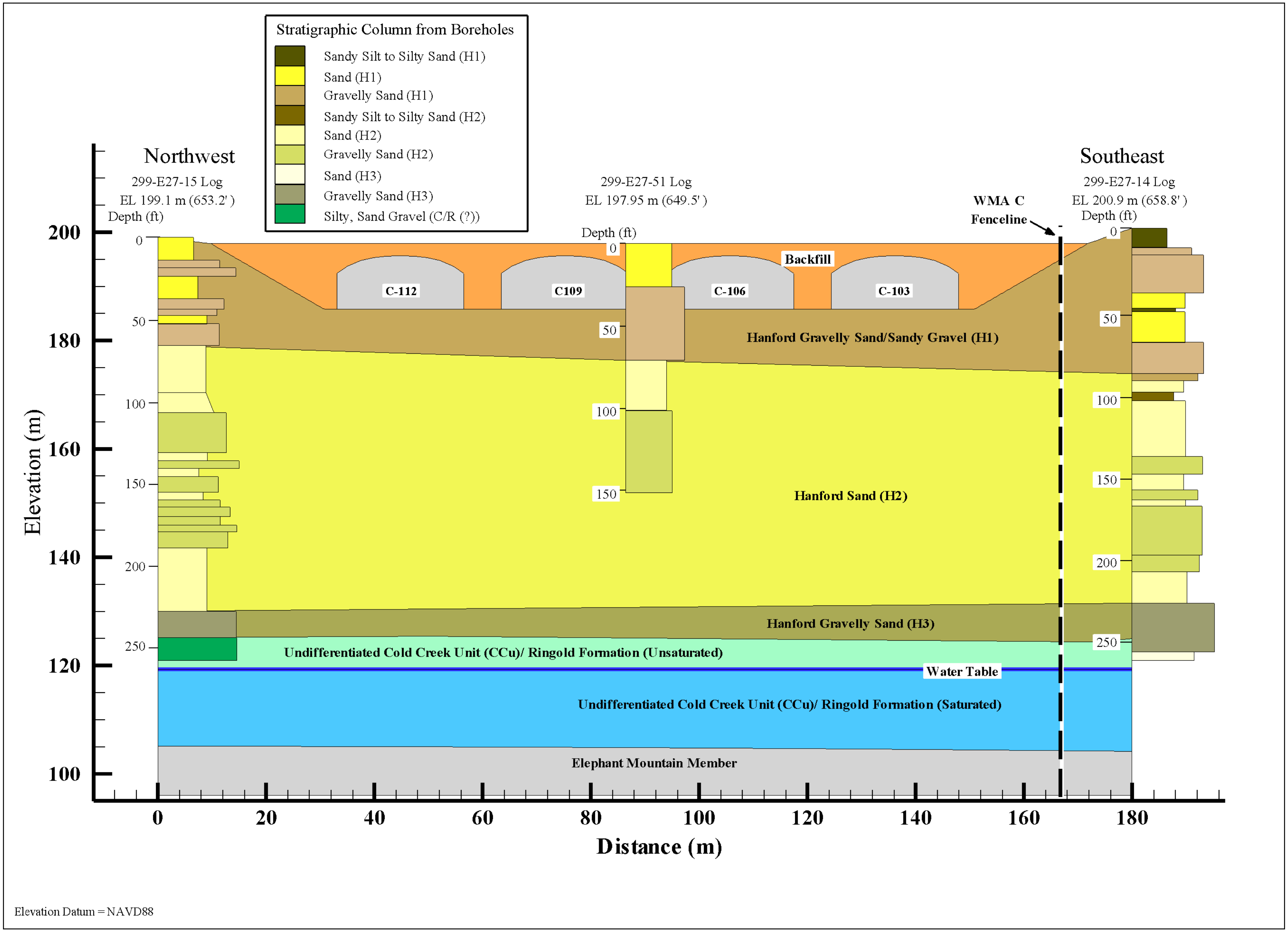
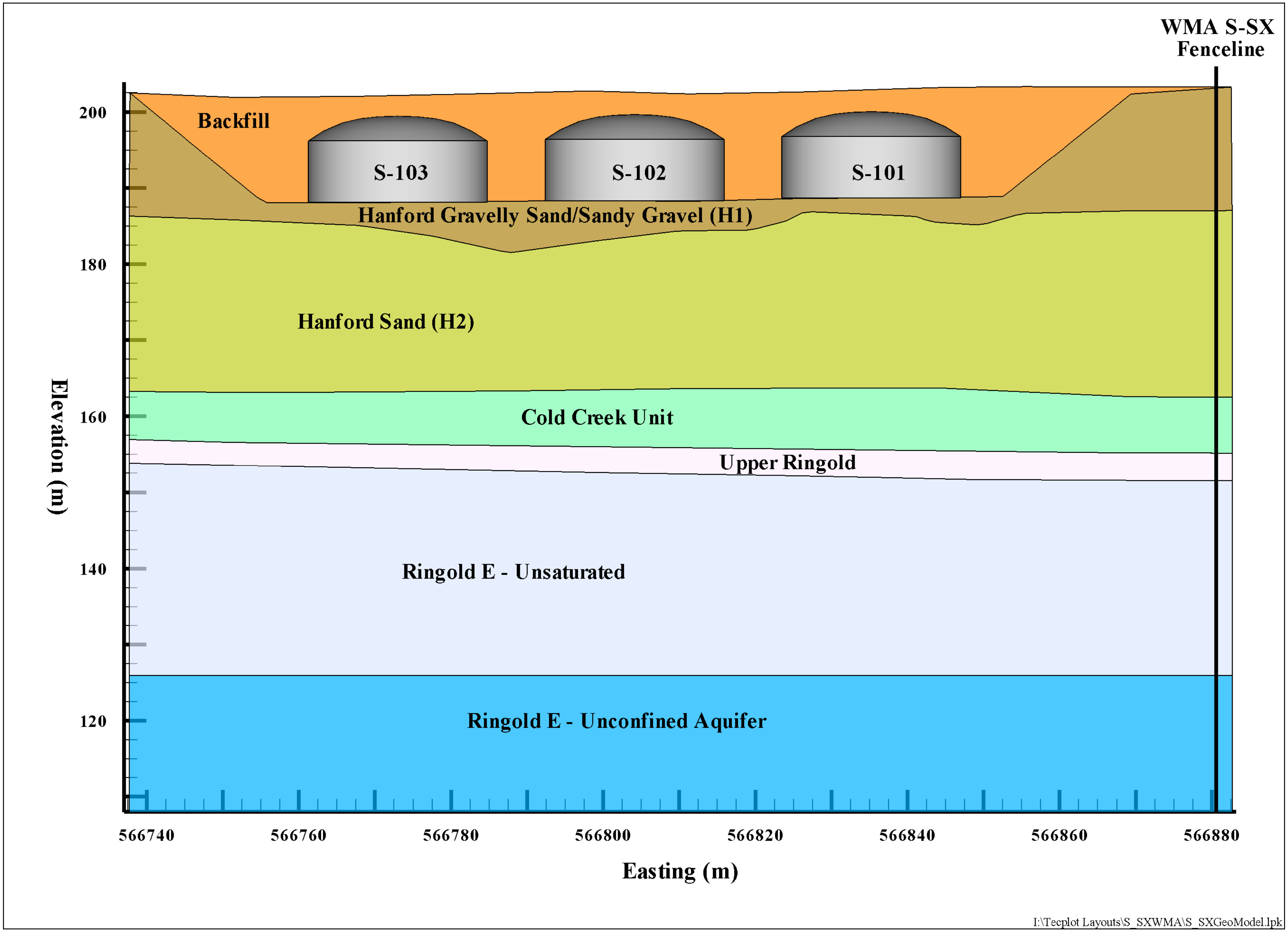


Figure 3-4. West-East Cross-Section through Waste Management Area S-SX



Despite its limitations, the empirical approach is considered adequate for several reasons. A range of K_d values can be derived for any contaminant discharged into the subsurface underlying the WMAs because contaminant reactivity with the subsurface system is dependent on the chemical nature of the contaminant and the ambient geochemical environment. If the geochemical environment remains fairly stable and can be simulated in the laboratory, a reproducible database can be developed to measure a range of K_d values that is reliable, regardless of the exact chemical reactions controlling observed behavior. Numerous analyses of undisturbed vadose zone soils and water chemistry at many locations in and around the WMAs have defined a consistent geochemical environment (Section 3.2.2.4.7). A long history of experimental work (Section 3.4.4.1.3) has provided an extensive database that has measured the reactivity of numerous contaminants under site-specific geochemical conditions characteristic of the ambient vadose zone. From this database, bounding ranges of K_d values have been developed for many contaminants of interest (Section 3.4.4.1.3).

Finally, for past releases and potential future leaks, the contaminant migration is assumed to be controlled by the current ambient geochemical environment. For past releases, the contaminant distribution in the soils was driven by the addition of tank waste into the vadose zone with chemical properties quite different from ambient soil water. Tank fluid properties (e.g., high salt content, high heat) influenced water and contaminant migration in the vadose zone near the source of release temporarily (Appendix D of Knepp 2002a). At the SX tank farm, for example, tank leaks containing hot, caustic, saline solution (8 to 10 molar sodium at 350°F or more) occurred. Cesium-137 mobility was thus greatly enhanced because high sodium content in the tank fluid pre-empted sorption sites causing cesium to migrate deeper than usual within the vadose zone (Appendix D of Knepp 2002a). Recent field characterization studies show that while these contaminants were mobilized shortly after the tank leak, their current state of mobility is consistent for ambient conditions. For example, desorption experiments and solids characterization data from recent characterization borehole sediments (e.g., Knepp 2002a) show that cesium-137, strontium-90, and uranium are now largely immobile in vadose zone soil. The potential for enhanced mobility for contaminants associated with past releases has been considered in these analyses through the use of effective K_d values associated with chemically impacted soils (pore water chemistry having high pH). See Section 3.4.4 for additional details on the geochemical model and parameters used for the flow and contaminant transport calculations.

3.2.2.4.6 Vadose Zone Flow and Transport Numerical Model. A two-dimensional flow and transport numerical model along a row of tanks was used for the integrated vadose zone-unconfined aquifer vertical cross-section. To account for three-dimensional aspects, the tank centerline mass flux and BTCs were transformed to average values across the tank farm fenceline on the basis of comparison of three- and two-dimensional results; the comparison evaluated the peak to peak comparison of contaminant concentrations for a long-lived mobile radionuclide (Zhang et al. 2004). See Section 3.2.2.4.9 for additional details.

The two-dimensional numerical model for WMA C assumes that the groundwater flow beneath the WMA is parallel to tank row C-103, C-106, C-109, and C-112. This flow direction is assumed to be consistent with the post-Hanford unconfined aquifer hydraulic gradient. Similarly, the two-dimensional numerical model for WMA C assumes that the groundwater flow beneath the WMA is parallel to tank row S-101, S-102, and S-103. This flow direction is assumed to be consistent with the post-Hanford unconfined aquifer hydraulic gradient.

As discussed earlier, for past releases, the vadose zone flow and transport model does not model the release event itself but uses the contaminant footprint as an initial condition for modeling past releases. A further discussion on the use of contaminant footprint in modeling is presented in Section 3.2.2.4.7.

The vadose zone simulations were composed of steady-flow and transient components, where flow fields developed from the steady-flow component were used to initialize the transient simulation. Steady-state initial conditions (that represent pre-Hanford Site operations) were developed by simulating from a unit hydraulic gradient condition to a steady-state condition, dictated by the initial meteoric recharge at the surface, water table elevation, water table gradient, no flux vertical boundaries, distribution of hydrologic properties, and location of impermeable tanks.

Simulations already completed in support of WMAs C and S-SX closure risk assessments are used as templates for other WMAs in the 200 East and 200 West Areas, respectively. That is, the relatively slight differences in geology between tank farms in a given part of the 200 East Area or 200 West Area are ignored. Also, this approach assumes that the post-closure groundwater flow direction for the other WMAs is predominately from west to east and parallel to the tank rows oriented in that direction. See Section 3.2.2.4.8 for a discussion on the justification for this approach. Future versions of this SST PA will perform vadose zone calculations based on site-specific data for other WMAs.

The steady-flow simulation, representing flow conditions for the year when a tank farm construction is completed (e.g., 1945 for C tank farm and 1952 for S tank farm), was used as the initial condition for all subsequent flow and transport simulations.

Transient conditions were conducted for the period from the time of tank farm construction to the year 2000, followed by a 10,000-year closure period (i.e., years 2032 to 12032) that involved changes in the flow fields in response to current conditions, placement of closure barrier, and effects of a degraded barrier. The infiltration (recharge) estimates for various times are described in Section 3.4.2.

An equivalent porous continuum model (Section 3.2.2.4.7) is assumed; fluid flow within the vadose zone is described by Richards' equation (Jury et al. 1991). The contaminant transport is described by the conventional advective-dispersive transport equation with an equilibrium linear sorption coefficient (K_d) formulation. A further justification for using a linear isotherm model is presented in Section 3.2.2.4.7. A series of mobile to moderately retarded contaminant species (K_d = 0, 0.02, 0.1, 0.2, 0.6, 1.0, 2.0, and 5.0 mL/g) were calculated for each run. The use of a suite of distribution coefficients allows for application of simulated results to a wide range of contaminants of concern (CoC). No temperature effects are considered for the vadose zone model (i.e., the model used is isothermal) (Section 3.2.2.4.7).

The vadose zone model considers the ubiquitous lateral flow in 200 Areas. As is evident from a large number of field observations in the 200 Areas: 1) lateral movement of water and contaminants is usually significant if the medium is stratified (as in the 200 Areas), 2) the initial moisture content is low, 3) the size of the application area is small relative to the size of the unsaturated zone, and 4) the application rate is small (Gelhar et al. 1985). Further details are provided in Section 3.2.2.4.7.

A key assumption in the modeling is that each of the three contaminant sources can be modeled separately and that temporal and spatial superposition can be used to estimate the cumulative impacts from different contaminant sources for each WMA. Detailed justification for superposition as well as for other assumptions used in modeling are presented in Section 3.2.2.4.7.

3.2.2.4.7 Justification for Flow and Transport Models Used. An understanding of the transport behavior of what has already leaked and how rapidly it is moving in vertical, as well as in lateral, directions within the vadose zone is useful in developing conceptual models for contaminant transport from all sources within a WMA. Based on extensive site characterization and field data, the vadose zone flow and transport model used in the SST PA incorporates a number of important characteristics. These include:

- Use of an equivalent porous continuum model
- Contaminant footprint as an initial condition for past releases
- Ubiquitous lateral flow
- Use of an isothermal model
- Use of a linear isotherm K_d model
- Use of superposition.

As discussed earlier, results from a separate task established a basis for the vadose zone conceptual model used in the SST PA. The results of this effort are described in Ye et al. (2005) and in Yeh et al. (2005). As discussed in Yeh et al. (2005), the simulated moisture plume does reproduce the general behavior of the observed moisture plume at the field site. Spatial moments of the simulated plume based on the effective hydraulic conductivities are in reasonably good agreement with those for the observed plume (Figure 3 of Yeh et al. 2005).

Use of an Equivalent Porous Continuum Model. To describe the bulk (or mean) flow behavior, each heterogeneous formation (e.g., gravelly sand unit in Figure 3-3) was replaced by its homogeneous equivalent, and effective or upscaled flow parameters were used to represent the homogeneous equivalent. Each formation unit was assigned different hydraulic properties. The laboratory-measured hydraulic properties were upscaled. Upscaling accounts for the fact that the numerical modeling applies to a scale that is much larger than the core scale at which laboratory measurements are available. As will be explained in Section 3.4.4.1.2, saturation-dependent anisotropy relationships (Polmann 1990) were invoked in recognition of field data from controlled and uncontrolled experiments that clearly show the dominant effect of lateral flow for the highly heterogeneous vadose sediments at the Hanford Site.

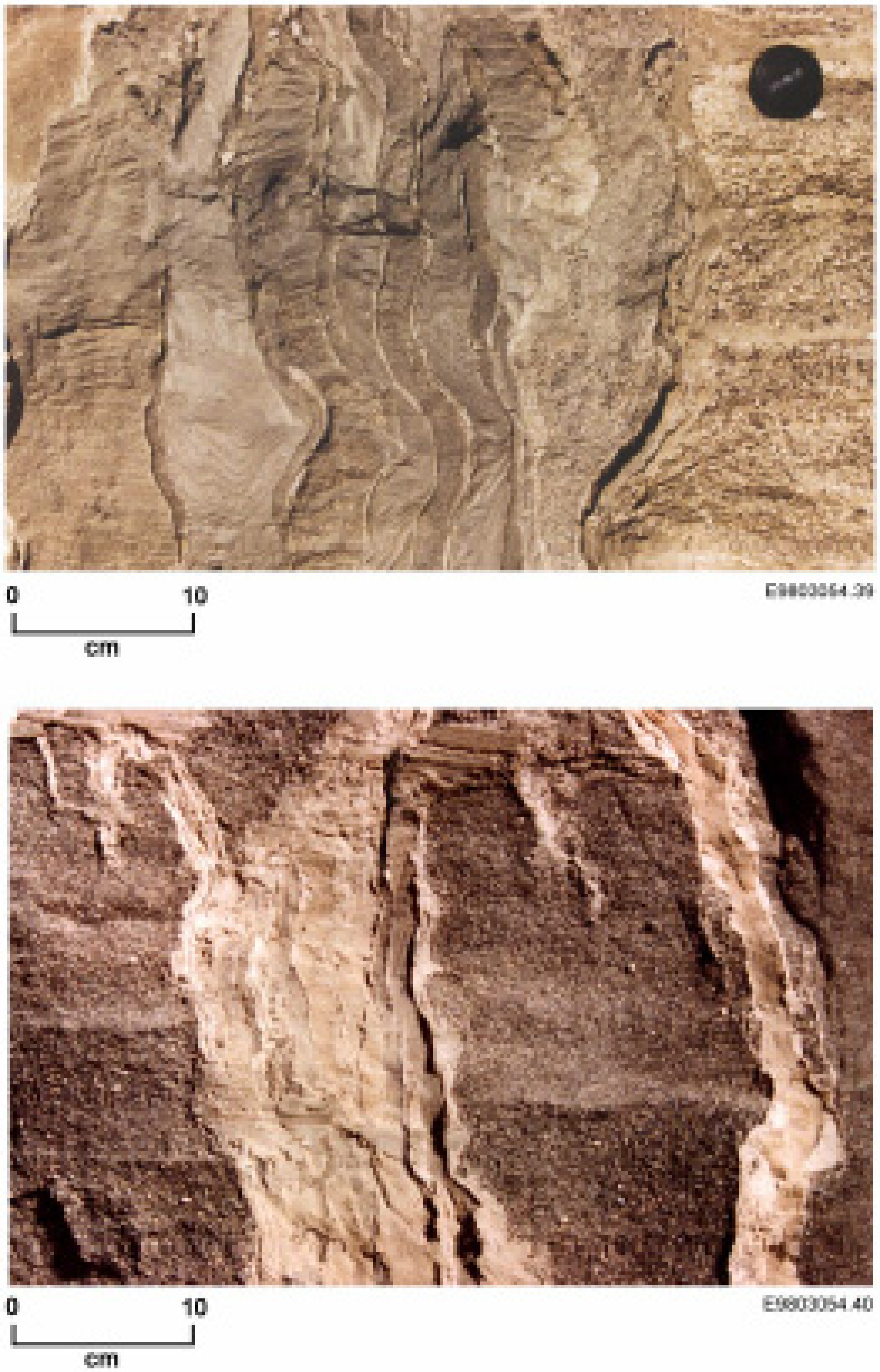
> Each heterogeneous geologic unit within the vadose zone is replaced by its homogeneous equivalent (see Figure 3-3 for WMA C and Figure 3-4 for WMA S-SX). Each geologic unit is assigned its upscaled hydraulic properties.

The vadose zone flow and transport simulations were based on the porous continuum modeling assumption. Such an assumption is supported by field data on moisture and contaminant plumes at various controlled and uncontrolled experiment sites. The most likely "fast flow" path in Hanford sediments is due to presence of clastic dikes (Figure 3-5) that often cross cut sedimentary units, especially in the Hanford formation. The dikes are ubiquitous sedimentary structures observed in outcrops and trenches that expose the Hanford formation in the 200 Areas (Fecht et al. 1999). These are believed to represent dewatering structures that developed during compaction and settling of cataclysmic flood deposits during or soon after floodwaters drained from the Pasco Basin. The dikes are of particular interest because they occur as near-vertical tubular bodies filled with multiple layers of unconsolidated sediments. There is very little evidence, however, to indicate that they extend all the way from near the ground surface to the water table.

In general, the hydraulic properties of clastic dikes can be considered essentially as a subset of the porous matrix properties for the Hanford sediments. This is based on laboratory measurements of clastic dike samples. In general, clastic dike sediments represent properties of fine sediments such as fine sand, silt, and clay, and can therefore represent regions of high moisture content (Murray et al. 2003). Under unsaturated flow conditions, however, the dikes can act as a barrier to flow rather than as fast flow channels. For example, if the clastic dikes were filled with gravelly sediments (with large pore sizes), it is *not* feasible, for the following reasons, to have a scenario under unsaturated conditions where the bulk of the flow is through the dikes.

- The porous matrix has a much smaller average pore size than the gravelly media within the clastic dike.
- For the moisture regime under low recharge conditions, the gravelly sediments with a larger pore size than the surrounding porous matrix will have a limited ability to hold moisture, and the fluid will be attracted primarily to the porous matrix. The conceptual model schematic in Figure 3-6, where the bulk of the flow bypasses the media with large pore sizes under unsaturated conditions, illustrates this scenario.

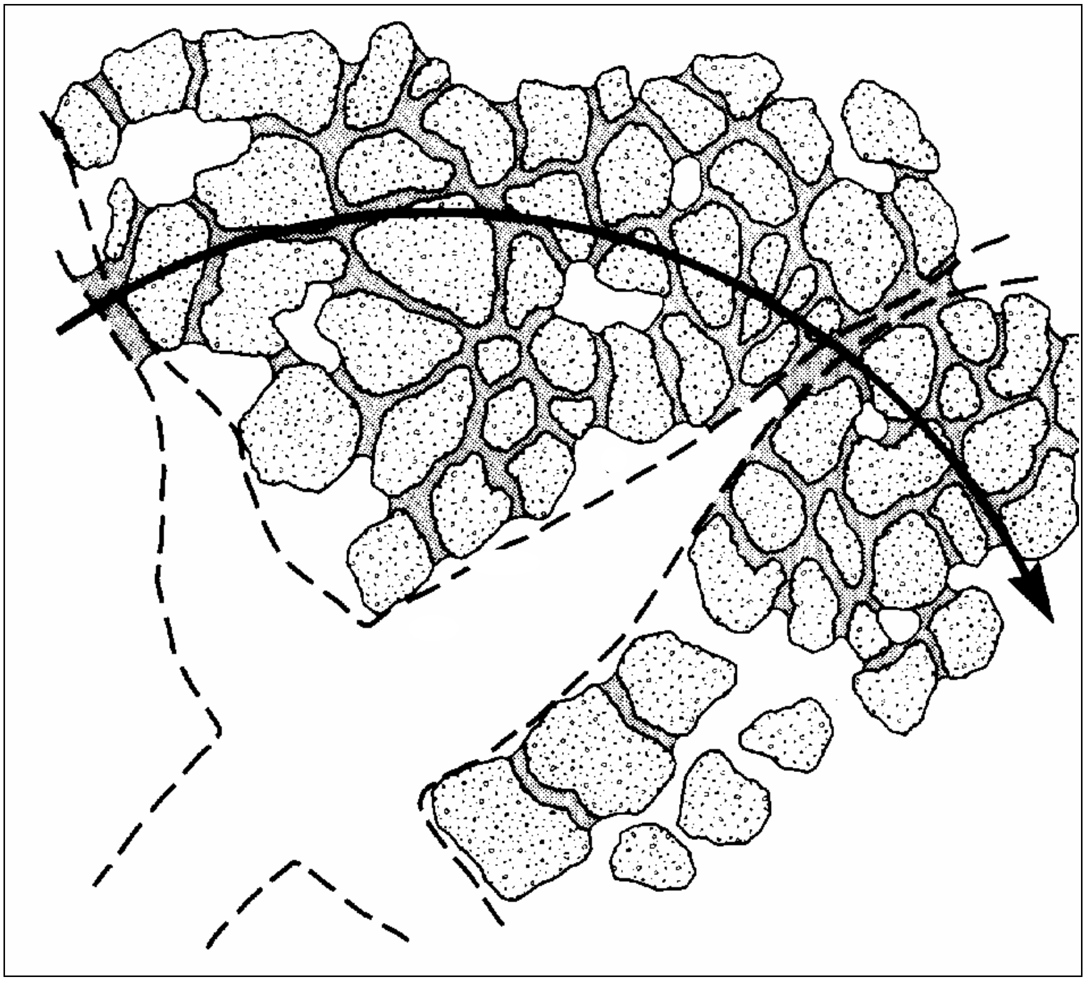
Figure 3-5. Infilled Sediments within Clastic Dikes [a]



[a] After (Fecht et al. 1999)

The middle portion of the two figures above show the infilled sediments within a dike; the host sediments are shown on the left and right edges of the two figures.

Figure 3-6. Conceptual Model of Fracture Flow under Unsaturated Conditions [a]



[a] After Wang and Narasimhan (1985)
The expanded vertical slice illustrates the fact that under unsaturated conditions and low recharge, the bulk flow bypasses the pathway formed by larger pore sizes and essentially follows the pathway formed by smaller pore size network. The large, open spaces in the figure mimic large pores such as those in a gravelly medium.

Thus, while clastic dikes do exist, it is less likely to intersect large segments of leaked wastes, and, when it does, the cross-sectional area of the intersection is small. Therefore, the presence of clastic dikes in unsaturated media appears unlikely to contribute much to the transport of the bulk quantity of leaked wastes and to long-term risk relative to higher peak concentrations for long-lived mobile radionuclides in groundwater. This is supported by the WMA S-SX FIR simulation results (Knepp 2002a).

The numerical results are also supported by studies reported elsewhere. These studies suggest that although preferential flow has been recognized and widely studied under saturated or near-saturated flow conditions (Nkedi-Kizza et al. 1983; De Smedt and Wierenga 1984), there is little evidence of it in arid and semiarid climates or under low water fluxes, particularly where soils are coarse-grained, such as those under the tank farms. Thus, under natural recharge conditions, precipitation at arid sites is usually too low (in relation to saturated hydraulic conductivity) to invoke preferential flow; much of the water in the dry soils is simply adsorbed onto the grain surfaces and cannot move along preferred pathways.

Contaminant Footprint as an Initial Condition for Past Releases. For past releases, a leak itself is not modeled; rather, a representative footprint of the vadose zone contamination was used as the initial condition for modeling liquid flow. At the SX tank farm, for example, tank leaks occurred that contained hot, caustic, saline solution of 8 to 10 molar sodium at 350°F or more. As discussed earlier, such chemical perturbations were most significant at the time of the leak event and shortly thereafter and at locations closest to the leak origin. However, the chemical buffering capacity of soil eventually overcomes tank chemistry influence, and releases are controlled by the ambient geochemical environment. The current far-field physical, thermal, and chemical behavior of the contamination footprint is considerably different from the near-field physical, thermal, and chemical behavior in the vicinity at the time of a tank leak and approaches conditions closer to the deeper, undisturbed vadose zone. The use of the contaminant footprint as an initial condition therefore provides an attractive alternative to long-term modeling of actual tank leak events in a WMA. Further details on use of such a modeling approach are presented in Section 3.2.2.4.5.

Ubiquitous Lateral Flow. The highly heterogeneous nature of Hanford sediments is indeed very effective in smearing out the effects of large natural or manmade applications. This is best illustrated by the moisture content profiles (Ye et al. 2005; Yeh et al. 2005) at the controlled field injection experiment (the Sisson and Lu site) in the 200 East Area in the vicinity of WMA C. The site was used for an infiltration test in the year 2000 (Gee and Ward 2001). Water content distribution was measured on May 5, 2000, at the 32 radially arranged cased boreholes. Injections began on June 1 and 4,000 L of water were metered into an injection point (point source) 5 m below the land surface over a 6-hour period. Similarly, 4,000 L of water were injected in each subsequent injection on June 8, June 15, June 22, and June 28. During the injection period, neutron logging in 32 wells took place within a day following each of the first four injections. A wildfire burned close to the test site and prevented immediate logging of the moisture content distribution for the fifth injection on June 28. Three additional readings of the 32 wells were subsequently completed on July 7, July 17, and July 31. During each neutron logging, water contents were monitored at 0.305-m (12-in.) depth intervals starting from a depth of 3.97 m and continuing to a depth of 16.78 m, resulting in a total of 1,344 measurements for the eight observation times over a 2-month period. The moisture content profiles, as shown in Figure 3-7, clearly illustrate significant lateral spreading. As indicated in Figure 3-8, the pre- and post-injection moisture plumes are essentially confined within three layers (i.e., two fine-textured layers and a coarse-textured layer that is sandwiched in between the two fine-textured layers). Such behavior of the moisture plume is related to the moisture-dependent anisotropy phenomenon (Ye et al. 2005; Yeh et al. 2005). Such field-scale processes are included in the modeling.

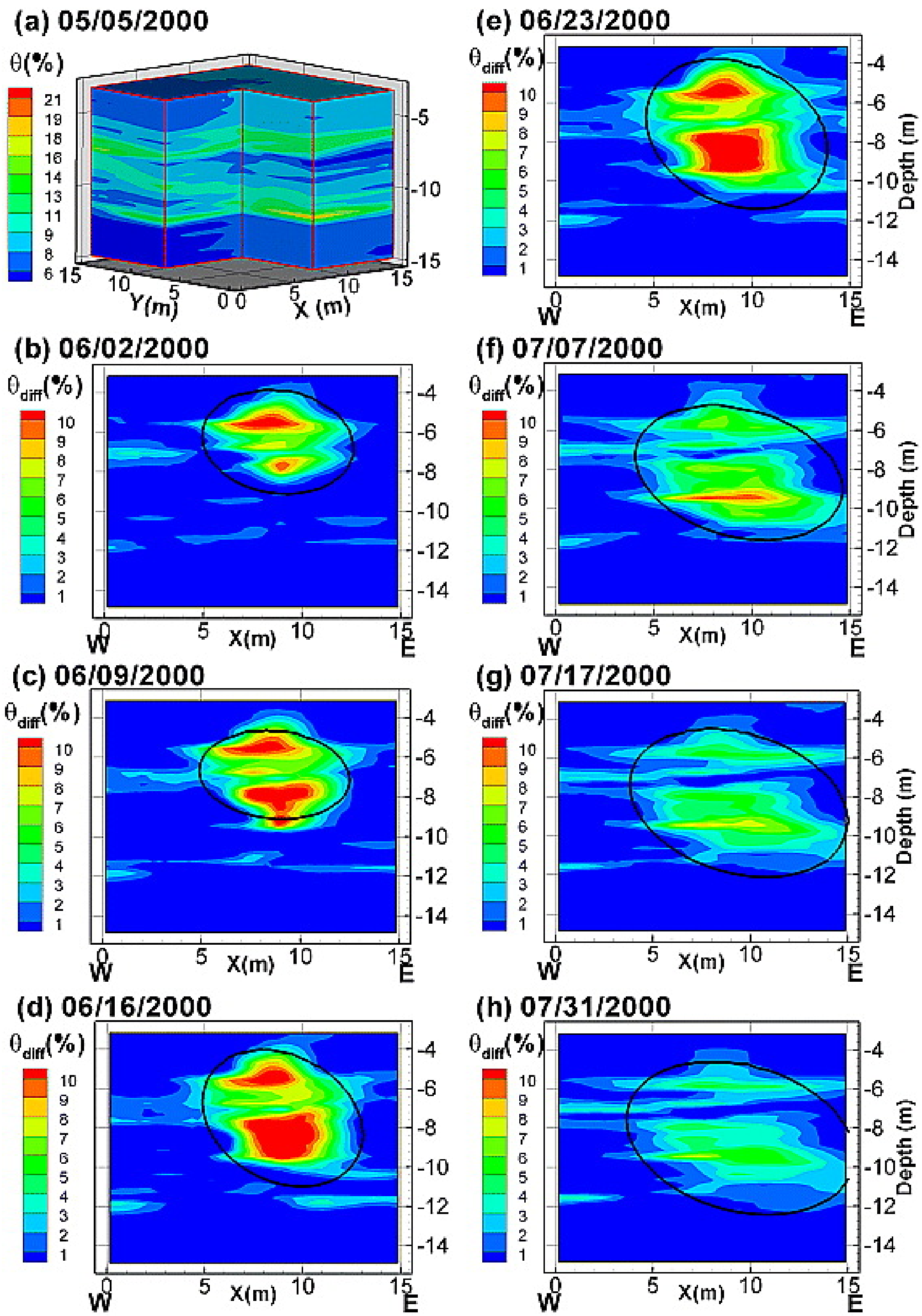
The preponderance of lateral migration is also evident elsewhere. The 241-T-106 tank leak (115,000 gal) is the largest known tank leak at the Hanford Site. The leak occurred in 1973 at a corner of the tank. Figure 3-9 shows the 1993 technetium-99 profile in borehole 299-W10-196 in the vicinity of the tank leak (Freeman-Pollard et al. 1994). The vadose zone profile clearly shows that even after 20 years of migration, the contaminant peak concentration for the long-lived mobile radionuclide is contained primarily within the fine-textured horizons at a depth of 35 to 40 m bgs and well above the water table. These field data suggest that the natural heterogeneity of the Hanford sediments plays an important role on flow and transport, and the significant lateral transport, which is in fact induced by media heterogeneities, is highly effective in containing plumes within the vadose zone for an extended period. A further corroboration of this phenomenon is evident at the 216-B-26 trench site in the 200 East Area. The BC cribs and trenches received nearly 30 Mgal of scavenged tank waste with possibly the largest inventory of technetium-99 ever disposed to soil at the Hanford Site. There is no evidence of groundwater contamination yet. In fact, field measurements suggest that, because of significant macroscopic anisotropy that is induced by media heterogeneities, the bulk of the technetium plume is concentrated within the fine-textured sediments at a depth of 30 to 35 m bgs almost 50 years after the high-volume discharge (Rucker and Sweeney 2004).

Thermal Effects and Use of an Isothermal Model. It should be noted that since the tanks contained large volumes of radioactive material, the heat generated by the decay of those radioactive materials heat the surrounding soil (Appendix D of Knepp 2002a). All simulations, nonetheless, were run using an isothermal model. The isothermal assumption is supported by a comparison of simulation results for non-isothermal and isothermal runs that appeared in Appendix D of Knepp (2002a).

Non-isothermal model simulations indicate that during periods of high-heat loads in the 1950s and 1960s, the thermal load from the boiling waste tanks altered flow patterns and caused large-scale redistribution of moisture. As a result, fluid and vapor flow near the high-heat tanks was dominated by vapor-liquid counterflow. Therefore, to understand the historical behavior, it is important to consider the strong coupling between the thermal and hydrologic environments. However, for impact assessment on the basis of the current thermal conditions in tank farms, long-term simulations on migration of long-lived mobile radionuclides were not significantly different for isothermal and non-isothermal conditions (Knepp 2002a).

Use of a Linear Isotherm K_d Model. As discussed earlier, no thermodynamically based conceptual or numerical models are presently available that are robust enough to accurately predict the degree of contaminant adsorption by undisturbed and disturbed sediments. The vadose zone conceptual model uses an empirical distribution coefficient, K_d, (or linear isotherm) to represent contaminant adsorption for relatively immobile contaminants (e.g., uranium). An inherent drawback of the K_d approach, of course, is its empirical nature. However, a considerable database of Hanford-specific K_d measurements is available (Krupka et al. 2004) for a variety of geochemical conditions and contaminants. For the expected concentrations of contaminants in the far field away from the near-field region of a tank leak, sorption can be considered to be independent of contaminant concentration and, therefore, K_d is assumed to be a constant for a given sediment-contaminant combination (Krupka et al. 2004). Therefore, given the extensive “empirical” database of site-specific conditions, the use of the K_d approach is considered to be a useful and valid approach for modeling contaminant adsorption for long-term risk assessments such as the SST PA. This approach has been used extensively in other risk assessments (Wood et al. 1995a, 1996; Mann et al. 2001; Knepp 2002a, 2002b).

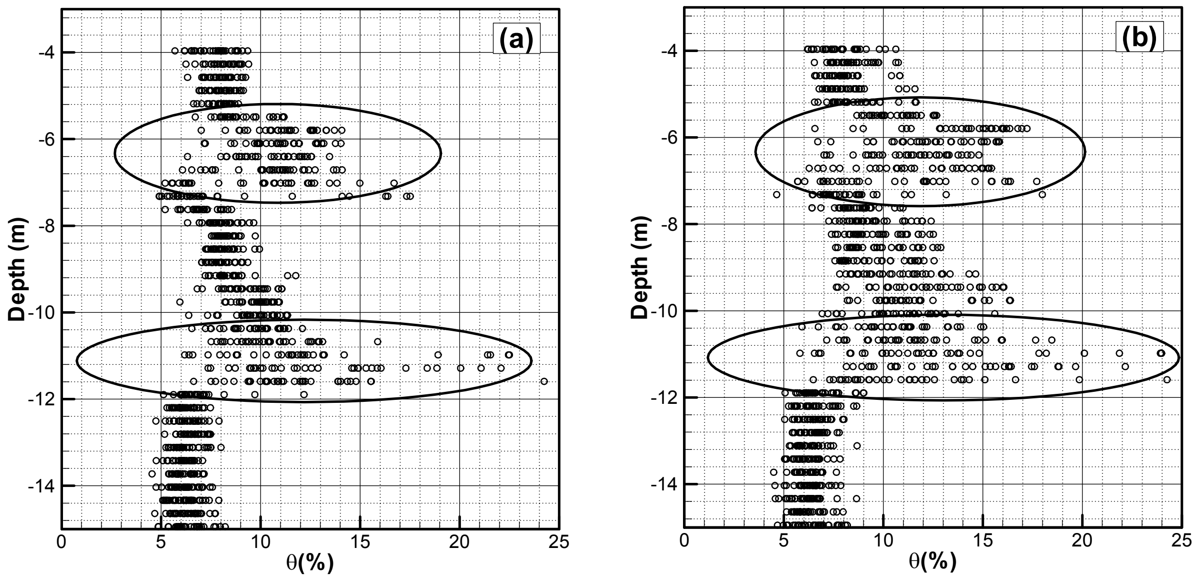
Figure 3-7. Moisture Content Profiles for the Field Injection Experiment in the 200 East Area [a]



[a] *After Ye et al. (2005)*

(a) Initial moisture content on May 5, 2000, and (b) through (h) are east-west trending cross-sectional views of moisture content (θ) differences (measured θ – initial θ) along the plane passing through the injection well. The solid curves are the fitted ellipsoids.

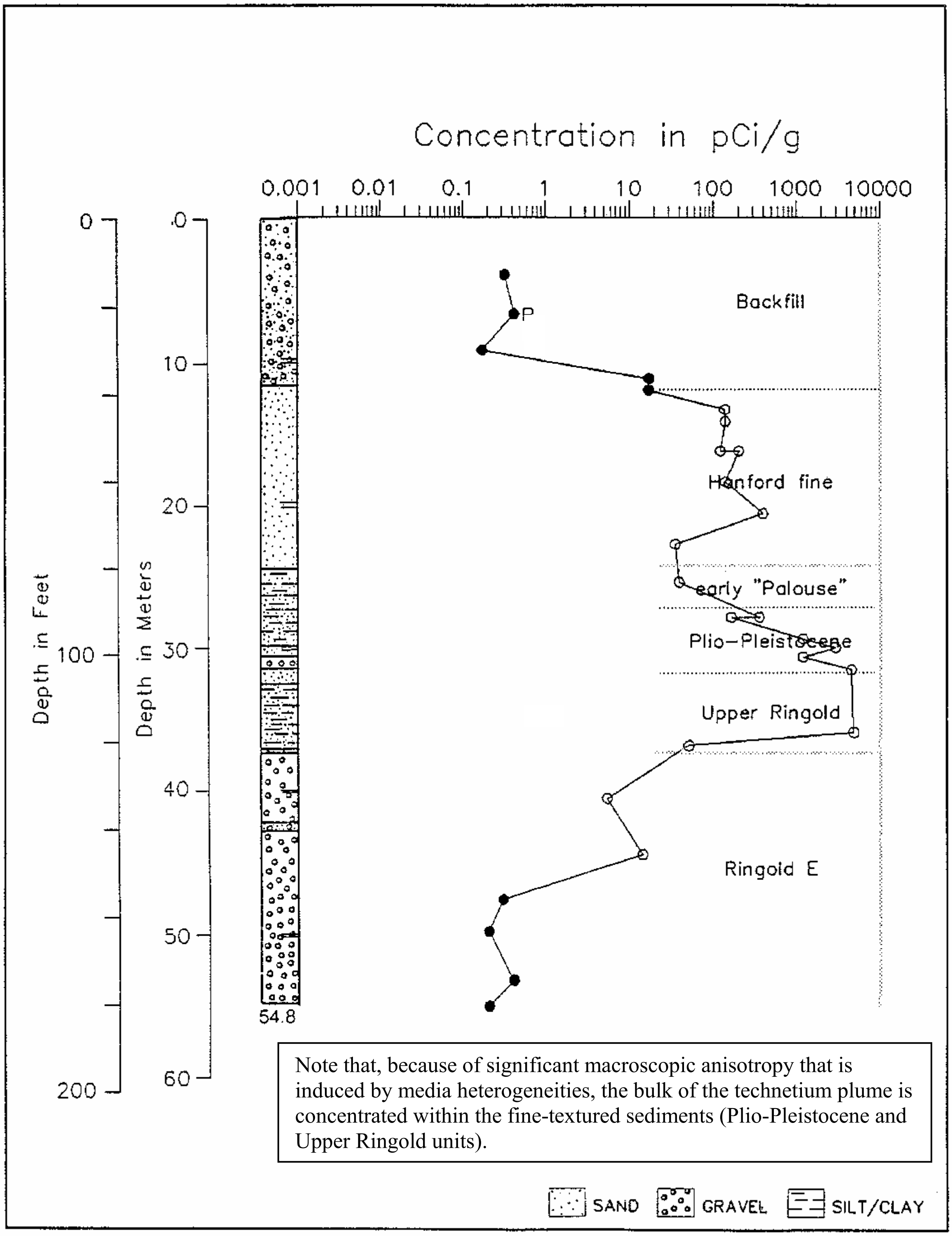
Figure 3-8. Pre- and Post-Injection Moisture Plumes for the Field Injection Experiment in the 200 East Area [a]



[a] After Ye et al. (2005)

Profiles of moisture content (%) measured on (a) May 5, 2000, and (b) July 31, 2000. The figures illustrate the fact that, in the absence of manmade injections, moisture contents at the field site are in equilibrium with natural recharge at the site.

Figure 3-9. Technetium-99 Profile in Borehole 299-W10-196 from 241-T-106 Tank Leak [a]



[a] *After Freeman-Pollard et al. (1994)*

Use of Superposition. A key assumption for the SST PA calculations is that each of the three contaminant sources can be modeled separately, and that temporal and spatial superposition can be used to estimate the cumulative impacts from different contaminant sources for each WMA. The assumption that different sources can be modeled separately implies that the contaminant release and transport for each source is independent from one another. This assumption is plausible because the past releases are modeled at locations beneath the tank and ancillary equipment waste residuals; the release models for these releases assume that the contaminants are available for transport with the infiltrating moisture. This assumption also assumes that the chemical adsorption of contaminants from one source is not impacted by another contaminant source within the WMA. This second assumption has not been substantiated. Geochemistry studies have indicated that high pH in the pore water can impact the K_d associated with contaminant transport through the vadose zone (Krupka et al. 2004). This potential impact needs to be addressed in future revisions to the SST PA. As a first approximation, the magnitude of this effect has been investigated in the sensitivity cases that look at minimum and maximum K_d values for CoCs (Section 3.5.3.4)

> For each contaminant source in a WMA, the principle of spatial and temporal superposition is used to obtain a composite contaminant breakthrough curves at the WMA fenceline for all sources (Section 3.2.2.4.7).

3.2.2.4.8 Extrapolation of WMA C and WMA S-SX Results to Other WMAs. As indicated earlier, the numerical simulations were not performed for all tank farm WMAs. Detailed simulations were conducted for all three source terms for WMA C and WMA S-SX in the 200 East and 200 West Areas, respectively. Results based on numerical simulations performed for WMA C were used as a template for other 200 East Area farms, and results from WMA S-SX were used as a template for results for other farms in 200 West Area. Specifically, the BTCs for WMA C and WMA S-SX provided inventory normalized concentrations in the groundwater at the WMA fenceline for each contaminant source (i.e., past releases, tank residuals, and ancillary equipment residuals). To estimate the concentrations for each tank row in any WMA, the appropriate inventories associated with the contaminant sources for that tank row were multiplied by the appropriate BTC for that WMA (200 East or 200 West Area) and source term to obtain the contaminant concentration for that source term and tank row.

This approach for estimating the impacts from other tank rows in other WMAs is considered reasonable at this time for the following reasons:

- The general stratigraphy in the 200 West Area is similar for other WMAs in that area. The general stratigraphy in the 200 East Area is similar for other WMAs in that area as depicted by the fence diagram for each WMA in Chapter 2.0. (A major impact on simulation is the lateral migration of the contaminant plume within the vadose zone.)
- The distance to the groundwater for each WMA in the 200 West Area is approximately the same; similarly, the distance to the groundwater for each WMA in the 200 East Area is approximately the same. (A major impact on simulation is transport time for contaminant plume to the aquifer.)

- The sediment-contaminant distribution coefficients (K_d) for soils on the 200 Area plateau appear to be independent of location. (A major impact is retardation of transport times for contaminants with non-zero K_d values.)
- The hydraulic properties of the 200 West Area aquifer are similar beneath each WMA in that area; similarly, the hydraulic properties of the 200 East Area aquifer are similar beneath each WMA in that area. (A major impact is dilution of the contaminant flux at downgradient locations.)

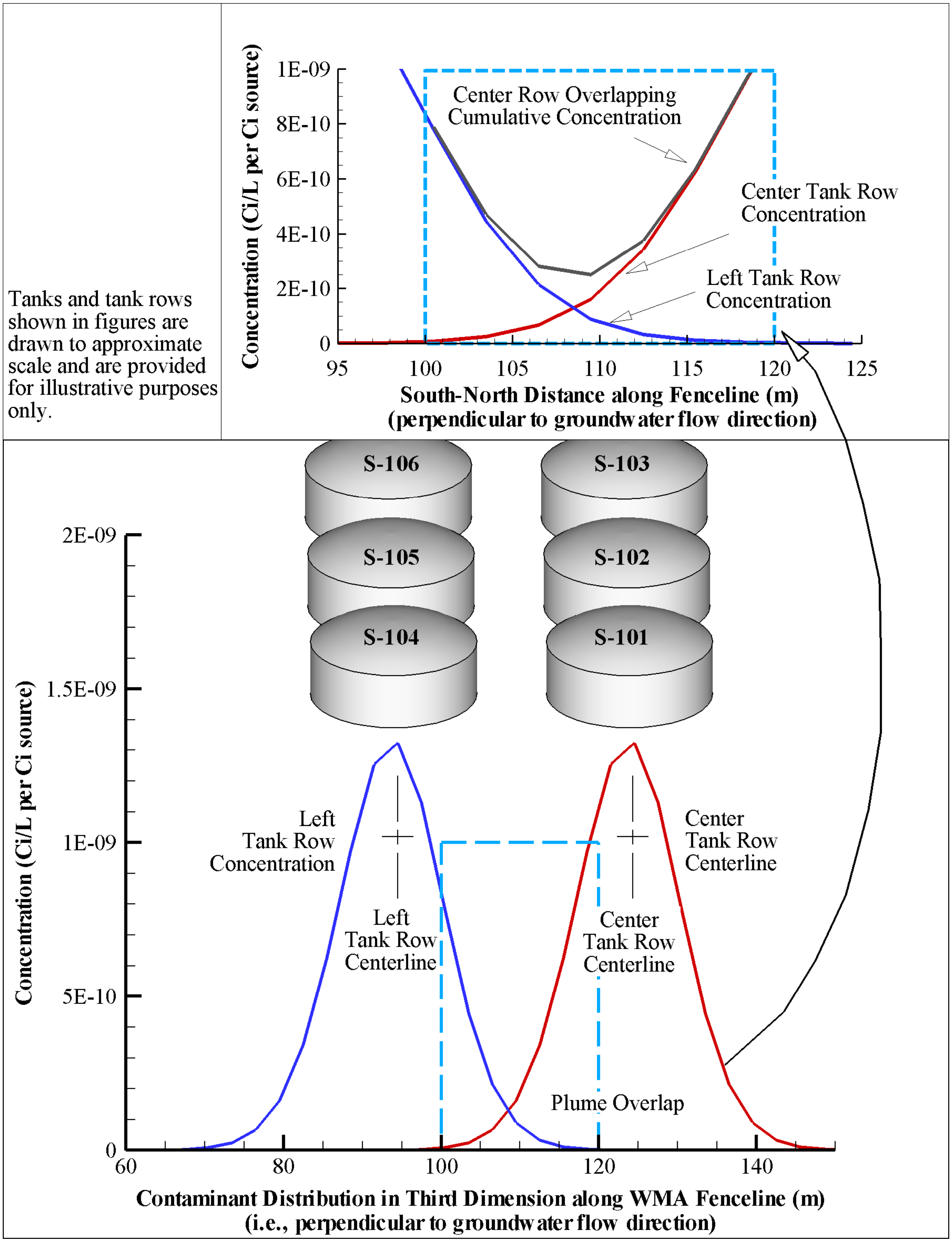
A major impact neglected in extrapolation of WMAs C and S-SX results to other WMAs is the orientation of the groundwater flow with respect to the conceptual models for different WMAs.

3.2.2.4.9 Accounting for the Third Dimension. The model used to describe flow and transport up to the fenceline is an integrated two-dimensional, saturated-unsaturated, vertical cross-section along a row of tanks. Although the simulations in this analysis are two-dimensional, in reality, flow and transport from any source type will occur in three dimensions. Because of the long simulation times, simulating three-dimensional processes is not insignificant. Although two-dimensional (x-z; with x = horizontal dimension and z = vertical dimension) simulations have shorter run times, the absence of flow and transport in the third (y) dimension translates into higher concentration predictions. Therefore, results from two-dimensional simulations need to be translated into equivalent values for a three-dimensional domain to better predict contaminant concentrations in the groundwater; such a relation is not known a priori.

To evaluate the impact of the third dimension, an identical two-dimensional case was simulated in three dimensions (Zhang et al. 2004). This case scenario involved a hypothetical leak of 4,000 gal at the lower-right corner of tank S-103 that began on the first day of the year 2000. The leak was set to last for 14 days (Khaleel et al. 2006b) and contained a unit release of technetium-99 and uranium-238. The main difference between the two-dimensional and three-dimensional simulations was the thickness of the simulation domain in the horizontal direction (y direction) perpendicular to the flow direction. In the two-dimensional simulation, a unit width (1 m) was used. In three-dimensional simulations, the width was 153 m discretized into 3-m units. Hence, water and contaminant migration occurred in the y direction for the three-dimensional simulation, whereas it was absent in the two-dimensional simulation.

To examine the relationship between the concentrations for the two simulations, the fenceline aqueous concentrations of technetium-99 and uranium-238 with K_d = 0.03 mL/g were evaluated along the y direction for the three-dimensional simulation when the peak concentrations occurred (Zhang et al. 2004). Figure 3-10 shows that the highest concentrations occurred at the centerline of tanks S-101, S-102, and S-103 (y = 125 m), and as the distance from the tank centerline increased, the concentrations decreased. The concentration along the y direction was nearly symmetrical along the centerline of the tank. Approximately 99.4% of technetium-99 and 98.3% of uranium-238 were within 20 m of the centerline of the tank (from 105 to 145 m). Therefore, co-mingling of plumes (Figure 3-10) between rows is assumed to be negligible because the results of the three-dimensional simulation indicated that 99% of the contaminants remained within 20 m of the plume centerline.

Figure 3-10. Results on Co-mingling of Plumes Based on Three-Dimensional Simulations [a]



[a] *After Zhang et al. (2004)*

For both technetium-99 and uranium-238, the shape of the BTCs from the two-dimensional simulations was very similar to those from the three-dimensional simulations (Zhang et al. 2004). The ratio of the two-dimensional (C_{2d}) and three-dimensional (C_{3d}) peak technetium-99 concentrations was 41.1. The C_{2d}/C_{3d} ratio for uranium-238 with $K_d = 0.03$ mL/g was 36.6. The arrival time of the of the technetium-99 peak concentration from the two-dimensional simulation was 14 years earlier than that from the three-dimensional simulation. For uranium-238 with $K_d = 0.03$ mL/g, the arrival times of the first and second peak concentrations from the two-dimensional simulation were respectively 36 and 342 years earlier than those from the three-dimensional simulation.

Based on the preceding results on a minimum co-mingling of BTCs for adjacent row of tanks and the presence of an adequate separation distance for BTCs from neighboring tank rows, no summation of BTCs at the WMA fenceline for all tank rows was necessary to obtain the composite curve. This is because the BTCs for individual rows tracked different flow lines. The fenceline BTC that produced the maximum concentration among all tank rows was thus used as the composite BTC for a given WMA at its fenceline.

3.2.2.5 Estimated Human Health Risks

Unit health effects factors provided in Rittmann (2004) were used to convert the predicted groundwater contaminant concentrations into the estimated impacts of interest for this analysis. A unit health effects factor is a scenario- and contaminant-specific factor that provides the health effects per unit contaminant concentration in groundwater (e.g., all-pathways farmer dose and ILCR per pCi/L for radionuclides, ILCR and HI per mg/L for non-carcinogenic chemicals). Formulas and data used in calculating the factors are presented in Rittmann (2004). The beta and photon emitter dose from groundwater was calculated using unit dose factors based on an exposure of 4 mrem/yr to the maximally impacted organ. These unit dose factors were derived by dividing the 4 mrem/yr dose by MCLs provided in EPA guidance (EPA 2000a). Values for each metric were calculated by first multiplying the predicted groundwater contaminant concentrations by the appropriate unit health effects factor and then summing the contributions from all contaminants that contribute to a particular metric. The calculations were performed with the use of an integrated computational software platform (DMT). A general description of the software platform is provided in Section 3.3.3. A detailed description is provided in *Tank Closure Project Decision Management Tool Systems Requirements Specification* (Watson 2005b).

3.2.3 Air Pathway

Gases and vapors could travel upward from the closed WMA facility through the soil to the ground surface. As downward water flow also drives gases and vapors down, the air pathway is maximized with minimum downward water movement. Thus, no water flow is considered in the calculations for the protection of air resources. The air emissions following closure are estimated using a simple model that provides an upper bound on the possible doses from tritium and carbon-14, and the possible emission rate of radon-222 at the ground surface above the waste.

The principal mechanism by which nuclides migrate from the waste to the ground surface is gaseous diffusion. The analysis in Appendix E of *Exposure Scenarios and Unit Factors for the*

Hanford Tank Waste Performance Assessment (Rittmann 2004) shows that convection mechanisms such as atmospheric pressure and temperature variations, wind, and rainfall have negligible secondary effects on the release of contaminants to the air.

The diffusion of radioactive gases such as tritium (hydrogen-3 as water vapor), carbon-14 (as carbon dioxide), and radon-222 (an inert gas) can be represented using Fick's Law of diffusion with a loss term for radioactive decay (Jury et al. 1991). The amount available for diffusion (i.e., the source concentration) changes with time due to the release mechanism for the contaminants from the waste form and radioactive decay. Two cases (one for tritium and carbon-14, the other for radon-222) must be considered because the performance objectives differ.

Because the estimated WMA closure inventories for tritium and carbon-14 are small, a bounding approach was used to estimate the air release doses for this risk assessment. Specifically, half the entire tritium and carbon-14 inventories for each WMA are released over a 1-year period, the first year after closure. The other half diffuses downward. This approach ignores diffusion from the waste that has been occurring during the past decades. A bounding approach avoids the task of defining release mechanisms and rates of progress through the overlying soils. The air pathway doses are calculated by multiplying the total inventories at WMA closure for tritium and carbon-14 by their corresponding unit release dose factors from Rittmann (2004), and summing the effective dose equivalent (EDE) from these two contaminants.

The radon-222 emanation rate from the ground surface is estimated using the diffusion equation derived in Appendix E of Rittmann (2004). This rate depends on the thickness of the waste, the depth of the soil cover, the assumed diffusivity of radon gas through the waste and soil cover, and the concentration of radium-226 in the waste. The radium-226 produces radon-222 by radioactive decay. The radium-226 is produced by the radioactive decay of curium-242, plutonium-238, uranium-238, uranium-234, and thorium-230. Because the radium-226 accumulates slowly with time, with most of it coming from the uranium-238 and uranium-234, the radium-226 concentration reaches its maximum value at times greater than 100,000 years after closure.

Because the estimated WMA closure inventories for the precursors of radon-222 are small, a bounding approach was used to estimate the air release rate for this SST PA. Specifically, the maximum concentration of radium-226 in the waste was used in the diffusion calculation. It was assumed that the uranium has not migrated appreciably from its initial location in the waste. Both the residual tank waste and the soil contamination plumes from tank leaks and other UPRs were assumed to be located 15 ft bgs. Note that such a depth is considerably shallower than what actually exists. In effect, the layer of grout and the tank dome are ignored. This approach certainly exaggerates the rate at which radon-222 escapes from the waste matrix and diffuses to the ground surface. The equation used to calculate the radon emanation rate at the ground surface is shown below:

$$J_{Rn}(z_0) = C_{0,Rn}\, U_{Rn}$$

$$\text{where,}\quad U_{Rn} = \frac{2\,\mu_{Rn} D}{1 - e^{-2\,z_0\,\mu_{Rn}}} \quad \text{and} \quad \mu_{Rn} = \sqrt{\frac{\lambda_{Rn}}{D}} \qquad \textbf{Eq. 3.7}$$

$$C_{0,Rn} = \frac{Q_{T,Ra}\,\lambda_{Rn}}{\lambda_{Rn} V + 2\,A\,\mu_{Rn} D\,\Omega} \quad \text{and} \quad \Omega = \frac{1 + e^{-2\,z_0\,\mu_{Rn}}}{1 - e^{-2\,z_0\,\mu_{Rn}}}$$

where:

A	=	surface area footprint of the waste disposal site, in m^2
$C_{0,Rn}$	=	average concentration of radon-222 in the waste pore space available for diffusion, in Ci per m^3 of waste
D	=	diffusion coefficient for radon moving through air trapped in the waste and soil, 0.01 cm^2/s = 31.56 m^2/yr
$J_{Rn}(z_0)$	=	upward diffusion flux of radon-222 at elevation z_0 above the waste, in Ci/m^2 per second
$Q_{T,Ra}$	=	the total quantity of radium-226 in the waste at a given time after closure, in Ci; it is assumed that the radium-226 and its ancestors are stationary in the waste
U_{Rn}	=	effective vertical diffusion speed of the radon-222 at the ground surface, in m/s
V	=	total volume of waste, in m^3
z_0	=	thickness of the soil above the waste, in m
λ_{Rn}	=	radioactive decay constant for radon-222, 66.21 per year
μ_{Rn}	=	inverse length parameter characteristic of radon-222 diffusing in the soil above the waste, 1.449 m^{-1}.

The radon diffusivity through the soil is taken to be 0.01 cm^2/s. This is based on the approximate binary diffusivity of radon in air (0.1 cm^2/s) scaled by a tortuosity factor of 0.1 to account for diffusion in the soil pore space.

The estimated impacts for these air releases are also discussed in Section 6.5.

3.2.4 Intruder Pathway

Two general cases of intruder exposures were evaluated. The first considers the radiation dose to an individual who excavates or drills a well into the closed WMA and brings some of the waste to the surface receiving an acute dose (contact with the waste for a relatively short period of time) (Section 5.3). The second considers the radiation dose to an individual who lives near the completed well receiving a chronic dose (exposed over a number of years) (Section 5.3).

Two acute cases were evaluated. The first involved excavating for a basement or building foundation or highway. The other acute case involved drilling a well through the buried waste. Because the WMA will be covered with at least a 15-ft soil surface barrier, the proposed excavations would not extend far enough below the ground surface to uncover any waste. The excavation scenario gives no radiation dose and is not evaluated any further. The construction of water wells in the 200 Areas is plausible due to the distance between the WMA and the nearest surface water (greater than 10 mi).

Three chronic cases were evaluated: 1) the rural farmer with a dairy cow, 2) the suburban resident with a garden, and 3) the commercial farmer. The chronic scenarios differ by what is done with the material taken from the well (well cuttings). The rural pasture scenario considers the well cuttings being scattered in a cow pasture. The suburban garden scenario considers a family planting a garden in the well cuttings. The commercial farm considers the well cuttings being present in an area that is planted with dry-land wheat, hay, or some other crop that is harvested and sold for profit. The owner of the commercial farm does not consume any of the crops himself. His only exposure to the exhumed waste occurs during the production of the crop.

The intruder analyses do not consider the effect of contaminated groundwater on the intruder by design (DOE 1999d). A complete evaluation of the exposure to the intruder would take into account the presence of mobile, long-lived radionuclides in the groundwater. However, because the time period of interest is 500 years after closure, it can be assumed that the waste contaminants have not migrated appreciably.

Groundwater pathway results are presented in Chapter 4.0. Thus, following current regulatory practices, the intruder analysis only evaluates the effect on the intruder from inadvertent contact with the exhumed waste and does not include exposure to groundwater.

The methodology used to assess the inadvertent intruder contaminant exposure for a closed SST WMA is based on the amount of contaminants and drilling spoils brought up during the drilling process and the assumptions associated with what is done with the material taken from the well. The amount of material taken from the well is directly proportional to the diameter of the well and the depth of the well (assumed to extend 6.1 m into the unconfined aquifer). Depending on location, a well could intercept only the residual tank waste or only the leak plume, or it could intercept both. Thus, three cases were considered for each of the significant contaminant sources for a given WMA:

- Tank residual only
- Tank leak only
- Tank residual and tank leak combined.

Leaks into the soil from other UPRs begin a short distance below the original ground surface and extend downward. Leaks into the soil during tank waste retrieval have not been included in the present analysis.

The fraction of the tank waste or soil contamination plume that is brought to the surface depends on the geometry of the waste. A cylindrical shape is assumed to represent the average waste distribution. For the underground tanks, the contaminated area is the entire tank bottom. The average waste thickness is about 1 in. and is assumed uniform across the tank bottom. The fraction of waste brought to the surface is calculated as the borehole cross-sectional area divided by the cross-sectional area for the tank.

For past releases, the geometric shape was assumed to be a cylinder with a vertical axis. The diameter and height of the cylinder are assumed to be equal to provide an average intruder case. The volume of the cylinder area was estimated from the volume of liquid leaked and an average soil filling fraction. The volumes of liquid were estimated from available historical records. The soil filling fraction was assumed to be 10%. Thus, about 10% of the soil volume is occupied by the aqueous waste that leaked. The fraction of waste brought to the surface was calculated as the borehole cross-sectional area divided by the cross-sectional area for the contaminated soil. Appendix E contains a more detailed discussion of the waste geometry models and gives the waste fraction brought to the surface for each waste form and intruder scenario.

In the scenario, not all of the waste material taken from the borehole is available for inhalation or ingestion by the various intruders. The particle size distribution of the cuttings typically includes larger pieces that cannot be inhaled or ingested. The large particles are consequences of drilling technology that breaks rocks only as much as needed to facilitate removal from the hole. This minimizes wear on the drill bit. In addition, the waste may be in a chemical form that resists uptake by plants or dissolution in lung fluid.

The wastes located in the UPRs are part of the soil. Therefore, 100% of the exhumed soil contamination is available for inhalation and ingestion by the intruders. However, the wastes located inside the tanks are attached to interior (iron) surfaces of the underground tank. The majority of this waste is located on the bottom of the tank, with grout on top of the waste. The chemical form of the residual tank waste is uncertain. It could be very hard and insoluble, and thus have reduced availability for inhalation and ingestion. After 500 years, the waste could also crumble readily during drilling and prove to be accessible for uptake into garden plants or pasture grass. The fraction available would be nearly 100% if the waste is in this form. In the present analysis, the fraction of the exhumed tank waste that is available for inhalation and ingestion is assumed to be 100%.

3.3 NUMERICAL IMPLEMENTATION

Numerical modeling was used to derive quantitative impacts from the proposed disposal action. The conceptual models discussed in the previous section were translated into numerical models. The numerical models were then implemented using computer simulations.

This section contains a description of the strategy used for the computer simulation and provides a summary of the selection criteria used for the computer code. A description of the codes and the criteria used in their selection, as well as the process of translating the disposal facility concepts and the natural system into computer models, is presented. The parameters used in the computer simulations are discussed in Section 3.4.

3.3.1 Code Selection and Verification

This section discusses the computer codes used for this SST PA and justifies their technical adequacy. Two major codes were used:

- Subsurface Transport Over Multiple Phases (STOMP)[1]
- Decision Management Tool (DMT).

Other codes were also used for simple data manipulation (Microsoft Excel[2] spreadsheets) and figure generation after processing of STOMP and DMT results (Tecplot[3]).

[1] Subsurface Transport Over Multiple Phases (STOMP) is copyrighted by Battelle Memorial Institute, 1996.

3.3.2 Subsurface Transport Over Multiple Phases

The computer code STOMP was chosen to model flow and transport through the vadose zone and groundwater out to the WMA fenceline. STOMP meets the requirements of *Computer Code Selection Criteria for Flow and Transport Code(s) To Be Used in Vadose Zone Calculations for Environmental Analyses in the Hanford Site's Central Plateau* (Mann et al. 1999) and has been used for a number of risk assessments on the Hanford Site (e.g., Knepp 2002a, 2002b). Verification of STOMP is documented in *CH2M HILL_STOMP Quality Assurance Test Report* (McMahon 2005a).

3.3.3 Decision Management Tool

The DMT software (*Tank Closure Project Decision Management Tool Functional Design Requirements* [Watson 2005a]) was used to calculate groundwater concentration of selected constituents from user-defined tank closure scenarios, predict risk associated with contaminant concentration, and compare predicted risk to regulatory criteria. Although the concentration and risk calculations used are not complex, the sheer volume of inventory and of fate and transport data used in these analyses makes hand calculations or spreadsheet use time- and cost-prohibitive. The DMT performs the defined risk assessment calculations in a more efficient manner and allows for greater sensitivity analysis to be performed than previously possible using spreadsheet calculations. The attributes of the DMT that make this possible include the following:

- Object-oriented data structures created from user-chosen input data in text files in the user's file system during program startup. In addition to maximizing calculation efficiency and minimizing computer resource usage, this also allows rapid response to new or altered input data, while giving the user the flexibility to add or remove input data without any programming knowledge.
- All necessary calculations are centralized in the source code and require no setup time on the part of the user.
- An intuitive user interface allows the user to create tank closure scenarios and calculate any of a number of provided risk metrics. The results can be examined numerically and graphically for each time step during the simulation both by analyte and cumulative risk.

Additionally, the DMT is a platform providing a means to distribute risk assessment data and results to multiple end users with better confidence in the accuracy of the results, while supplying complete traceability of input data and assumptions. Verification of the DMT is documented in *Tank Closure Project Decision Management Tool Software Test Report* (Watson 2005c).

[2] Microsoft and Excel are registered trademarks or trademarks of Microsoft Corporation in the United States and/or other countries.
[3] Tecplot is a registered trademark or trademark of Tecplot, Inc.

3.4 VALUES AND ASSUMPTIONS

This section describes and justifies the key data and assumptions used to estimate human health impacts associated with the tank closure system. This section also describes the selection criteria and key assumptions for the data used in the numerical models described in Section 3.2 for the barriers and key features impacting contaminant migration for different pathways.

The performance analysis examined three contaminant pathways: groundwater, air, and the inadvertent intruder. Controlling parameters associated with the operating features and processes are identified in Section 1.4.3. The barriers and key features impacting contaminant transport for the groundwater pathway are the surface cover, the grouted tank structure and ancillary equipment, the vadose zone, and the unconfined aquifer. Only the surface barrier and grouted tank structure and ancillary equipment are the barriers and key features impacting contaminant transport for the air and inadvertent intruder pathways.

Finally, this section defines the selection of the parameters associated with the reference case. (The definition of sensitivity cases and their associated parameter selection are discussed in Section 3.5.) This section has been organized into the following subsections:

- Inventory (Section 3.4.1)
- Surface cover and pre- and post-barrier recharge rates (Section 3.4.2)
- Grouted tank and ancillary equipment (Section 3.4.3)
- Vadose zone (Section 3.4.4)
- Unconfined aquifer (Section 3.4.5)
- Exposure scenarios (Section 3.4.6)
- Reference case (Section 3.4.7).

3.4.1 Inventory

Inventory remaining within the WMA subsurface at the time of closure is a key parameter for PAs because the estimated human health risks are proportional to the inventory of key contaminants associated with a closure site. This section provides an overview of the approaches used to derive the various inventory estimates and enabling assumptions. Detailed discussions of the methodologies used to develop the inventory estimates are available in cited references; inventory data are summarized in Appendix C.

3.4.1.1 Overview

Three sources of contaminants were considered in this SST PA:

- Past SST leaks and discharges of tank waste inside the tank farms
- Residual waste remaining in tanks after retrieval
- Residual waste in tank farm infrastructure (e.g., waste pipelines and catch tanks).

Inventory estimates for past tank leaks, discharges of waste to the soil column, and waste inventories associated with ancillary equipment were developed following the general methodology used to estimate current tank waste inventory estimates (Kupfer et al. 1998). That is, best estimates of waste volumes are coupled with projected waste compositions to develop inventory estimates. Inventory estimates associated with future activities such as waste residuals remaining in the tank after retrieval were developed using the HTWOS (Kirkbride et al. 2005). The HTWOS model combines current estimates of tank waste compositions from the BBI with projected retrieval methodologies to project waste residual compositions expected to remain in the Hanford Site waste tanks at the end of the retrieval process. The model includes experimental based wash and caustic leach factors to estimate the sluicing efficiency in removing sludge from the tanks. The model also provides composition estimates of fluids being removed from the tanks. Inventories from potential waste loss events during waste retrieval activities were developed by multiplying projected retrieval fluid compositions by the estimated leak volumes for each tank.

Inventory estimates used in this SST PA were taken from three primary documents. The latest assessment of SST leak volumes and waste loss volumes within the tank farms is documented in Field and Jones (2005). Inventory estimates for these leaks and waste loss events were reported by Corbin et al. (2005). Waste residuals remaining in the tanks after completion of retrieval activities were documented in Kirkbride et al. (2005). The inventory estimates for ancillary equipment (e.g., pipelines and MUSTs) within the tank farms are documented in Appendix C.

Inventory estimates in Appendix C include 25 chemicals, 46 radionuclides, and supplemental analytes as provided by HTWOS. However, previous risk assessment analyses have led to the conclusion that long-term risks are driven by a small subset of chemicals and radionuclides. For brevity, the data and discussions presented in this section focus on this major contaminant subset. All contaminants listed in Appendix C were included in the modeling analysis.

3.4.1.2 Past Leaks

Human health impacts of leaks from SSTs or other waste loss events within the SST farms are closely linked to both the type of waste and the volumes of waste lost to the soil column. Inventory estimates for chemicals and radionuclides lost to the vadose zone were developed by integrating information from historical tank farm records with recent field investigations data. Historical tank farm records include compilations of waste transfer records (Anderson 1990; Agnew 1997), extensive documentation of process wastes being transferred to the SSTs (see process chemistry discussion in Appendix B), and analysis of the historical gross gamma logging data. Field data include recent spectral gamma logging of all drywells in all SST farms (Field and Jones 2005), as well as results from selected drilling and sampling projects (Knepp 2002a, 2002b). The general approach used to develop SST leak inventories is outlined schematically in Figure 3-11 and is documented in *Inventory Estimates for Single-Shell Tank Leaks in S and SX Tank Farms* (Jones et al. 2000b). The time and volumes of SST leaks were developed based on an analysis of historical records and field characterization data (Field and Jones 2005). The development of leak inventory estimates is documented in Corbin et al. (2005). Inventory data were developed for each waste loss event that has been identified within each tank farm (Corbin et al. 2005) and are available in the cited report. The complete suite of inventory estimates for past leaks is documented in Appendix C of this SST PA.

Inventory estimates associated with past leaks from tanks, UPRs, tank residuals, and ancillary equipment within WMAs C and S-SX are provided in Table 3-1 for selected radionuclides and chemicals. Contaminants listed in Table 3-1 for the groundwater pathway will be among the first to arrive in the unconfined aquifer beneath each WMA and, as shown by past analyses (Mann et al. 2001; Lee 2004), will dominate the post-closure groundwater pathway impacts. Contaminants listed for inadvertent intruder impacts are the contaminants shown by past analyses (Mann et al. 2001; Mann and Connelly 2003) to be the major contaminants contributing to the intruder doses. Complete inventories for all past tank leaks, UPRs, tank residuals, and releases from ancillary equipment for these two WMAs and other WMAs are provided in Appendix C.

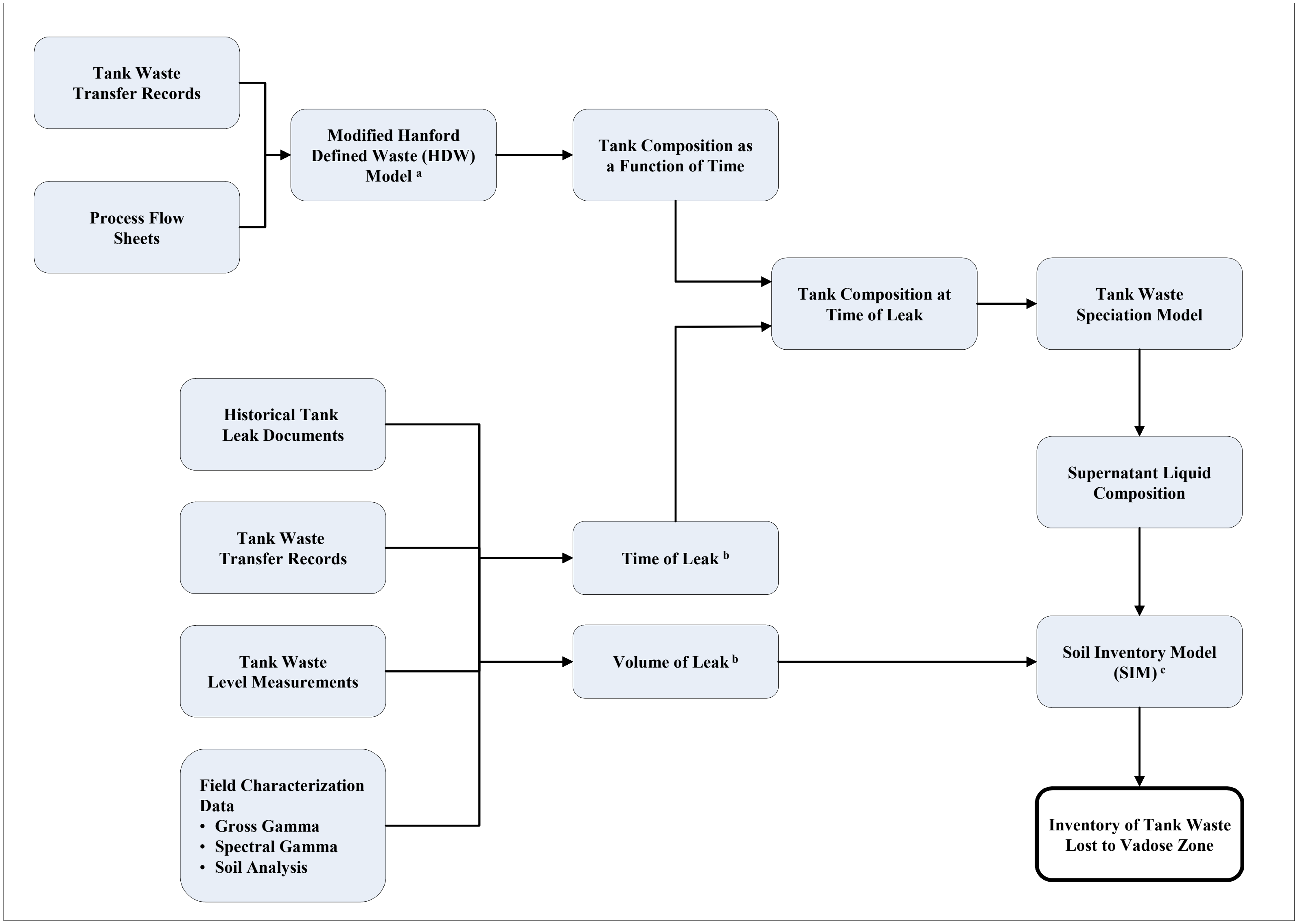
3.4.1.3 Tank Residual Wastes

Estimates for the residual waste inventory in each SST were developed as part of the HTWOS model run (Kirkbride et al. 2005). The general approach used to develop tank residual waste inventory estimates is outlined schematically in Figure 3-12. Further discussion of the HTWOS model is provided in Sections 2.5.1.1 and 2.5.3. As discussed in Section 2.5.3, the HTWOS model assumed 360 ft^3 of waste remained in the 100-Series SSTs and 30 ft^3 remained in the 200-Series SSTs. The composition of the residual waste in the 100-Series SSTs was estimated as 35 wt% water washed solids with one-half concentration of bulk as-retrieved supernate (except tank C-106, which uses current BBI since it has been retrieved). Values for the wash and caustic leach factors are based on experimental data (reported in the BBI).

Table 3-1 summarizes the inventory for residual waste in each SST for WMAs C and S-SX. Complete inventories for all contaminants within the residual waste for each tank are provided in Appendix C.

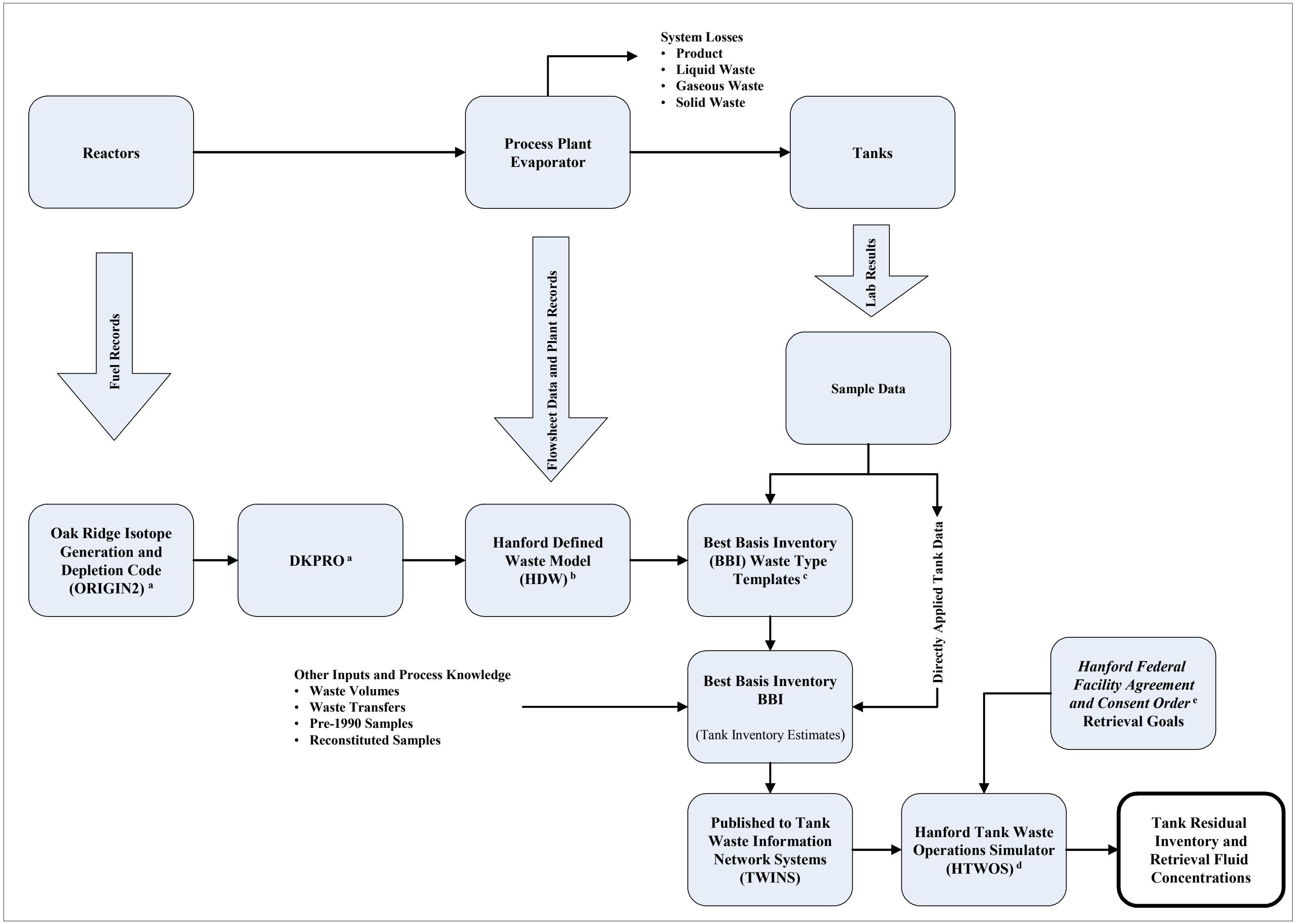
> A complete list of contaminant inventory for each WMA and for all contaminants is presented in Appendix C. Inventory estimates for tank waste residuals as well as those for ancillary equipment residuals are based on HTWOS. Inventory estimates for past tank leaks as well as UPRs are based on SIM.

Figure 3-11. Process for Determination of Inventories from Past Leaks



[a] *Higley and Place (2004)*
[b] *Field and Jones (2005)*
[c] *Corbin et al. (2005)*

Figure 3-12. Process for Determination of Inventories from Tank Residuals



[a] *Watrous (2002)*
[b] *Higley and Place (2004)*
[c] *Place et al. (2005)*
[d] *Kirkbride et al. (2005)*
[e] *Ecology et al. (1989)*

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Waste Management Area C												
Tank Row 241-CR Vault												
Past Releases												
UPR-200-E-81	2.74E-02	2.38E-02	2.17E+01	5.81E+03	1.78E+03	7.34E+01	2.74E-02	2.72E-04	8.60E+01	6.88E-01	1.62E-01	7.59E-01
UPR-200-E-82	1.42E+00	8.39E-04	1.62E+01	7.64E+02	3.93E+02	2.44E+01	1.42E+00	1.84E-02	1.48E+02	4.96E-02	1.15E-02	6.62E-02
UPR-200-E-86	4.92E+00	2.61E-03	6.04E+01	1.27E+03	1.49E+03	1.68E+02	4.92E+00	6.51E-02	1.98E+04	3.44E-01	7.49E-02	4.57E-01
Other Residuals												
241-CR vault	4.60E-02	2.18E-04	1.68E+00	4.42E+01	1.76E+01	2.88E+03	4.60E-02	9.01E-03	3.40E+02	3.38E+00	7.26E-01	3.88E+00
Ancillary Equipment (pipelines)												
Plugged and blocked pipelines	3.58E-05	3.03E-07	1.05E-02	3.13E+00	7.13E-02	7.78E-08	3.58E-05	1.36E-07	1.58E-10	6.03E-06	2.03E-06	5.92E-04
Tank Row C-101/C-104/C-107/C-110												
Past Releases												
C-101	2.25E-01	1.21E-04	1.51E+00	5.03E+01	4.07E+01	7.75E+00	2.25E-01	2.96E-03	8.52E+02	1.90E-02	3.87E-03	2.10E-02
C-104	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
C-107	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
C-110	2.43E-02	3.11E-05	1.47E+00	1.30E+03	5.08E+01	1.63E+01	2.43E-02	2.70E-04	7.45E+01	2.50E-02	2.46E-03	1.07E-02
UPR-200-E-107	7.07E-05	9.06E-08	3.14E-03	3.44E+00	1.39E-01	4.69E-02	7.07E-05	7.65E-07	2.17E-01	5.60E-05	6.04E-06	3.10E-05
Tank Residuals												
C-101	2.79E-03	1.68E-03	4.18E+00	3.37E+02	3.84E+01	6.41E+02	2.79E-03	7.80E-05	2.44E+02	8.07E+00	1.43E+00	1.43E+00
C-104	2.35E-01	4.23E-03	9.75E+00	7.40E+01	1.38E+02	3.12E+03	2.35E-01	5.33E-04	4.62E+02	3.63E+01	9.33E+00	4.44E+01
C-107	2.13E-01	2.72E-03	6.25E+00	3.34E+02	1.34E+02	1.76E+04	2.13E-01	5.28E-01	3.98E+02	1.85E+01	3.00E+00	5.68E+01
C-110	1.67E-01	1.18E-06	7.83E+00	5.62E+02	3.39E+01	9.08E+01	1.67E-01	1.13E-04	2.07E+02	1.99E+00	2.17E-01	1.20E+00

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Tank Row C-102/C-105/C-108/C-111												
Past Releases												
C-102	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
C-105	2.26E-01	8.85E-05	1.42E+00	1.14E+02	3.77E+01	8.99E+00	2.26E-01	3.04E-03	6.21E+02	1.60E-02	4.25E-03	1.88E-02
C-108	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
C-111	5.38E-02	2.64E-03	5.27E+00	1.07E+03	4.66E+02	8.42E+02	5.38E-02	6.95E-04	1.95E+02	1.00E-01	2.11E-02	2.49E+00
Tank Residuals												
C-102	3.20E-03	1.58E-03	5.83E+00	1.99E+02	5.78E+01	2.52E+02	3.20E-03	8.70E-05	1.21E+02	3.26E+01	7.81E+00	9.55E+00
C-105	5.57E-01	5.99E-04	2.56E+00	5.24E+01	4.34E+01	3.34E+03	5.57E-01	2.05E-05	5.71E+02	1.50E+01	2.96E+00	8.01E+00
C-108	4.05E-02	7.03E-06	5.92E+00	1.16E+02	6.43E+01	3.54E+02	4.05E-02	2.82E-04	3.51E+03	1.43E-01	1.55E-02	4.60E-01
C-111	1.58E-02	2.00E-04	2.19E+00	1.27E+02	6.58E+01	2.80E+04	1.58E-02	3.20E-04	1.22E+02	5.16E+00	1.00E+00	5.88E+00
Tank Row C-103/C-106/C-109/C-112												
Tank Residuals												
C-103	7.17E-02	4.63E-04	6.02E+00	5.81E+00	5.94E+01	2.48E+04	7.17E-02	2.06E-03	6.53E+02	4.26E+01	8.92E+00	1.94E+01
C-106 [e]	1.65E-01	6.30E-04	3.78E+00	3.48E+01	4.14E+01	6.61E+04	1.65E-01	1.62E+00	1.45E+03	1.67E+01	3.57E+00	6.52E+01
C-109	4.96E-01	4.28E-04	1.68E+00	1.61E+02	1.03E+02	8.45E+03	4.96E-01	4.44E-04	5.74E+03	2.17E+00	3.72E-01	1.67E+00
C-112	7.23E-01	3.41E-04	1.84E+00	2.45E+02	1.74E+02	1.72E+04	7.23E-01	4.72E-04	6.57E+03	2.39E+00	3.01E-01	1.34E+01
Other Residuals (Ancillary Equipment)												
C-301	1.25E-02	5.92E-05	4.57E-01	1.20E+01	4.77E+00	7.83E+02	1.25E-02	2.45E-03	9.22E+01	9.18E-01	1.97E-01	1.05E+00
Tank Row C-201/C-202/C-203/C-204												
Past Releases												
C-201	1.07E-02	6.42E-07	7.80E-01	9.69E+01	2.63E+01	1.91E+02	1.07E-02	1.42E-04	4.30E+01	1.17E-02	2.53E-03	5.56E-01
C-202	8.77E-03	5.25E-07	6.37E-01	7.92E+01	2.15E+01	1.56E+02	8.77E-03	1.16E-04	3.52E+01	9.56E-03	2.06E-03	4.55E-01
C-203	8.29E-03	4.96E-07	6.03E-01	7.49E+01	2.03E+01	1.48E+02	8.29E-03	1.10E-04	3.32E+01	9.04E-03	1.95E-03	4.30E-01
C-204	7.04E-03	4.21E-07	5.12E-01	6.36E+01	1.72E+01	1.26E+02	7.04E-03	9.33E-05	2.82E+01	7.68E-03	1.66E-03	3.65E-01

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Tank Residuals												
C-201	1.21E-02	6.88E-07	8.66E+00	8.60E+01	3.04E+01	1.08E+02	1.21E-02	1.53E-04	1.26E+01	8.95E+00	1.93E+00	1.42E+00
C-202	1.13E-02	6.66E-07	6.32E+00	7.77E+01	2.54E+01	2.04E+02	1.13E-02	1.48E-04	1.08E+01	7.90E+00	1.71E+00	6.67E-01
C-203	1.88E-03	6.61E-07	1.69E+01	1.60E+02	7.96E+00	2.78E+01	1.88E-03	1.47E-04	2.02E+00	1.55E+00	3.34E-01	7.88E-02
C-204	6.02E-03	5.31E-07	9.86E+00	3.13E+01	2.17E+01	1.52E+01	6.02E-03	1.18E-04	6.62E+00	7.99E-03	1.73E-03	1.95E-03
Waste Management Area S-SX												
Tank Row S-101/S-102/ S-103												
Tank Residuals												
S-101	6.77E-02	8.41E-04	6.19E+01	1.20E+02	4.14E+01	3.23E+03	6.77E-02	7.63E-03	1.43E+02	2.42E+00	4.75E-01	1.63E+00
S-102	5.87E-01	1.84E-04	5.90E+02	4.74E+02	9.52E+01	1.05E+04	5.87E-01	2.88E-01	6.13E+03	1.11E+01	2.38E+00	2.08E+01
S-103	7.10E-01	1.42E-04	7.07E+02	3.16E+02	7.22E+01	5.73E+03	7.10E-01	2.09E-01	1.70E+02	1.37E+01	2.86E+00	1.98E+01
Tank Row S-104/S-105/S-106												
Past Releases												
S-104	3.95E-02	5.57E-05	1.45E+01	4.99E+03	1.20E+03	1.02E+02	3.95E-02	3.68E-04	1.18E+02	4.61E-01	1.01E-01	6.11E-01
S-105	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
S-106	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Tank Residuals												
S-104	2.80E-02	2.84E-04	7.54E+00	1.38E+02	1.48E+01	2.78E+03	2.80E-02	3.28E-03	8.21E+01	2.45E+00	5.00E-01	1.42E+00
S-105	2.15E+00	1.71E-04	3.26E+02	7.08E+02	1.50E+01	3.45E+02	2.15E+00	7.26E-01	1.29E+03	1.70E+00	3.41E-01	1.37E+00
S-106	7.27E-01	2.20E-04	7.26E+02	9.09E+02	6.69E+01	1.85E+03	7.27E-01	1.90E-01	7.59E+02	1.02E+00	2.23E-01	3.30E+00
Tank Row S-107/S-108 /S-109												
Tank Residuals												
S-107	9.77E-03	8.97E-05	2.26E+01	2.92E+01	1.54E+01	1.24E+03	9.77E-03	1.23E-03	1.83E+02	5.08E+00	1.06E+00	3.13E+00
S-108	7.00E-01	2.12E-04	6.66E+02	7.91E+02	1.05E+02	3.13E+03	7.00E-01	1.94E-01	7.88E+02	4.50E+00	9.63E-01	1.25E+01
S-109	1.21E+00	2.00E-04	4.12E+02	1.63E+03	2.01E+01	4.93E+03	1.21E+00	4.15E-01	3.06E+01	4.99E+00	9.82E-01	4.07E+00

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Tank Row S-110/S-111/ S-112												
Tank Residuals												
S-110	2.11E-01	1.59E-04	9.69E+01	6.52E+02	3.83E+01	2.96E+03	2.11E-01	3.42E-02	1.24E+02	4.00E+00	7.90E-01	3.91E+00
S-111	3.28E-01	2.22E-04	1.60E+02	5.97E+02	8.37E+01	7.87E+03	3.28E-01	4.02E-02	2.06E+03	2.84E-01	5.90E-02	6.24E-01
S-112	4.84E-03	1.05E-03	1.82E+01	1.34E+01	3.54E+00	1.76E+04	4.84E-03	7.36E-01	2.45E+01	8.87E+00	1.57E+00	1.46E+01
Tank Row SX-101/SX-102/SX-103												
Tank Residuals												
SX-101	1.93E-01	1.86E-04	1.18E+03	9.32E+02	7.67E+01	1.93E+04	1.93E-01	1.14E-01	1.20E+03	2.86E+01	5.83E+00	3.14E+01
SX-102	9.80E-01	2.22E-04	6.35E+02	6.44E+02	2.04E+02	1.18E+04	9.80E-01	1.29E-01	3.79E+02	8.15E+00	1.71E+00	1.82E+01
SX-103	2.09E-01	1.88E-04	1.79E+02	6.93E+02	1.24E+02	6.32E+03	2.09E-01	3.94E-02	2.79E+02	3.12E+00	6.38E-01	8.96E+00
Other Residuals (Ancillary Equipment)												
SX-302	5.40E-03	5.87E-06	5.99E+00	1.10E+01	1.85E+00	6.13E+02	5.40E-03	1.09E-03	1.41E+01	7.24E-01	1.59E-01	6.41E-01
Tank Row SX-104/SX-105/SX-106												
Past Releases												
SX-104	4.51E+00	4.39E-03	6.17E+01	4.65E+03	2.46E+03	5.76E+01	4.51E+00	5.65E-02	5.93E+03	1.09E-01	2.38E-02	1.49E-01
SX-105	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
SX-106	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Tank Residuals												
SX-104	1.36E-01	9.23E-05	6.69E+01	3.30E+02	5.13E+01	3.42E+03	1.36E-01	1.85E-02	1.80E+02	4.23E+00	8.31E-01	3.07E+00
SX-105	4.27E-01	2.15E-04	2.76E+02	5.95E+02	1.90E+02	8.84E+03	4.27E-01	9.40E-02	5.49E+02	3.59E+01	7.01E+00	2.24E+01
SX-106	7.33E-01	2.89E-04	5.60E+02	6.99E+02	2.22E+02	1.65E+03	7.33E-01	1.78E-01	2.05E+03	1.24E+01	2.67E+00	2.21E+01

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Tank Row SX-107/SX-108/SX-109												
Past Releases												
SX-107	6.01E+00	9.28E-03	1.63E+02	1.23E+04	5.93E+03	1.41E+02	6.01E+00	6.57E-02	1.79E+04	2.88E-01	6.13E-02	3.82E-01
SX-108	1.40E+01	2.17E-02	3.81E+02	2.87E+04	1.38E+04	3.29E+02	1.40E+01	1.53E-01	4.18E+04	6.73E-01	1.43E-01	8.92E-01
SX-109	8.01E-01	1.24E-03	2.18E+01	1.64E+03	7.90E+02	1.88E+01	8.01E-01	8.76E-03	2.39E+03	3.84E-02	8.17E-03	5.10E-02
Tank Residuals												
SX-107	1.52E-02	1.65E-04	8.33E+00	6.96E+01	1.42E+01	3.13E+03	1.52E-02	1.73E-03	1.68E+02	3.89E+00	7.56E-01	1.94E+00
SX-108	7.16E-02	5.08E-04	8.78E+01	1.21E+03	5.07E+01	7.47E+04	7.16E-02	5.04E-03	2.11E+03	5.56E+01	1.08E+01	2.81E+01
SX-109	6.01E-02	6.88E-04	1.78E+01	2.82E+02	1.69E+01	1.77E+03	6.01E-02	7.51E-03	3.32E+02	2.42E+00	4.80E-01	1.65E+00
Tank Row SX-110/SX-111/SX-112												
Past Releases												
SX-110	3.35E-01	2.59E-04	4.43E+00	2.86E+02	1.17E+02	1.82E+01	3.35E-01	4.34E-03	1.42E+02	1.38E-02	3.14E-03	2.91E-02
SX-111	2.28E-01	1.56E-04	2.82E+00	1.44E+02	7.05E+01	7.86E+00	2.28E-01	2.95E-03	5.26E+01	9.01E-03	2.11E-03	1.52E-02
SX-112	4.01E-01	6.19E-04	1.09E+01	8.21E+02	3.95E+02	9.39E+00	4.01E-01	4.38E-03	1.19E+03	1.92E-02	4.09E-03	2.55E-02
Tank Residuals												
SX-110	4.70E-02	4.96E-05	1.58E+01	2.42E+02	2.27E+01	3.08E+03	4.70E-02	5.28E-03	6.32E+01	3.87E+00	7.57E-01	2.24E+00
SX-111	2.58E-02	2.70E-05	1.11E+01	1.18E+02	1.82E+01	3.29E+03	2.58E-02	3.14E-03	4.49E+01	4.12E+00	8.03E-01	2.21E+00
SX-112	2.10E-02	2.14E-05	1.01E+01	9.55E+01	1.68E+01	3.37E+03	2.10E-02	2.54E-03	3.93E+01	4.20E+00	8.17E-01	2.21E+00

Table 3-1. Inventories for Selected Rows within Waste Management Areas C and S-SX (6 pages)

Source [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 C [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Tank Row SX-113/SX-114/SX-115												
Past Releases												
SX-113	1.50E+00	2.39E-03	1.62E+02	7.87E+03	1.99E+03	1.40E+02	1.50E+00	1.59E-02	4.23E+03	2.88E-01	5.59E-02	3.80E-01
SX-114	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
SX-115	4.53E+00	7.04E-03	2.37E+02	1.43E+04	6.04E+03	3.11E+02	4.53E+00	4.88E-02	1.49E+04	9.56E-01	2.08E-01	1.27E+00
Tank Residuals												
SX-113	7.52E-04	7.70E-07	4.38E-01	3.90E+00	1.10E+00	1.27E+02	7.52E-04	5.12E-06	1.43E+02	3.55E-01	6.89E-02	2.27E-01
SX-114	5.87E-02	5.92E-04	1.64E+01	2.57E+02	2.49E+01	2.05E+03	5.87E-02	6.40E-03	3.22E+02	2.74E+00	5.39E-01	1.72E+00
SX-115	4.25E-02	4.21E-05	2.92E+02	2.71E-08	1.71E+00	2.00E+05	4.25E-02	5.25E-03	1.68E+01	2.35E+02	5.51E+01	2.12E+02
Ancillary Equipment (Pipelines)												
Plugged and blocked pipelines	1.77E-02	1.65E-05	2.23E-01	1.71E+01	8.46E+00	2.09E-01	1.77E-02	3.18E-04	2.11E+01	3.92E-04	7.60E-05	6.37E-04

[a] Major contaminants for groundwater impacts based on estimated chemical distribution coefficients ≤ 0.2 mL/g (Section 3.4.4.1.3).

[b] Major contaminants for inadvertent intruder impacts based on major contaminants contributing to the intruder doses in previous analysis (Mann et al. 2001; Mann and Connelly 2003).

[c] Tank residuals inventories from Kirkbride et al. (2005), inventories for tank leaks and unplanned releases from Corbin et al. (2005), pipeline inventories from Lambert (2005).

[d] Radionuclide inventories in Ci decayed to January 1 2004; chemical inventories in kg.

[e] Tank inventory for C-106 from best basis inventory (TWINS 2005).

3.4.1.4 Other Residual Wastes

The development of waste inventory estimates for ancillary equipment and transfer piping systems in the SST farms is still in its infancy. As discussed in Section 2.5.4, the scope of ancillary equipment included in this SST PA includes inactive MUSTs, vaults, and blocked and plugged waste transfer pipelines.

> A complete list of ancillary equipment associated with each WMA is provided in Section 2.4.3.

Transfer lines associated with the SST system are believed to be about 154 km (96 mi) in total length (Field 2003b). This includes approximately 1,400 different lines ranging in size from 2 to 6 in. in diameter. The average diameter is about 3 in. These transfer lines were directly contaminated with waste. Contaminant concentrations for the waste in the blocked and plugged pipelines were assumed to be that of the average contaminant concentration currently in the particular tank farm. To obtain the contaminant inventory, the presumed volume was multiplied by the estimated contaminant concentration (Lambert 2005).

3.4.1.5 Data Structure

As discussed in the conceptual model defined in Section 3.2.2.1, the SST PA analysis used the inventory data, among other inputs, to generate a contaminant BTC at the WMA fenceline. For example, in the C tank farm, the vadose zone contamination sources associated with the tank row containing tanks C-103, C-106, C-109, and C-112 were used to compute the BTCs based on the STOMP calculations, the DMT tool, and the superposition principle to estimate the contaminant concentrations in the groundwater at the WMA fenceline. This process was followed throughout the seven WMAs.

3.4.2 Surface Barrier and Pre- and Post-Barrier Recharge Rates

The primary feature of the surface barrier with respect to the groundwater pathway is to limit moisture infiltration into each closed WMA system. Long-term estimates are needed for the moisture flux through a fully functional surface barrier as well as for the degraded barrier. These estimates were derived from lysimeter and tracer measurements combined with a modeling analysis.

Results from more than three decades of work are available on meteoric recharge estimates at the Hanford Site. Infiltration (recharge) can vary greatly depending on factors such as climate, vegetation, surface condition, and soil texture. Studies conducted over the last decade at the Hanford Site (Gee et al. 1992, 1996; Wing and Gee 1994; Fayer and Walters 1995; Fayer et al. 1996, 1999; Ward et al. 1997) suggest that recharge rates can vary from less than 0.1 mm/yr (0.004 in./yr) on a variety of soil and vegetative combinations to greater than 130 mm/yr (5.1 in./yr) on bare basalt outcrops or bare, gravel-covered waste sites (Gee et al. 1992). Detailed experimental work has also been performed on infiltration rates through surface barriers (Fayer and Szecsody 2004).

For the SST PA analyses, recharge estimates were needed for four different time periods: 1) prior to construction of the tanks, 2) current operations (from now until the placement of a surface barrier), 3) surface barrier performance (during its design life), and 4) surface barrier performance for the degraded surface barrier (after its design life). The numerical simulations for the SST PA were run starting in the year tank farms were constructed, and continue for 10,000 years after closure. Recharge estimates for the period prior to the tank construction were needed to estimate the steady-state moisture conditions within the vadose zone prior to the start of simulations. Recharge estimates corresponding to current operations and corresponding to surface barrier performance are key inputs into the transport calculations discussed in Section 3.2.2.4.

3.4.2.1 Recharge Estimates for Pre-Development Conditions

Recharge estimates for the conditions prior to tank construction are based on correlations of soil types and infiltration characteristics of the native soils. Data supporting these recharge estimates for the 200 East and 200 West Areas soils are documented in Last et al. (2004b). Within the 200 East Area, recharge estimates range between 0.9 and 3.0 mm/yr for soils with established shrub-steppe vegetation. Similarly, within the 200 West Area, recharge estimates range between 3 and 4 mm/yr for soils with established shrub-steppe vegetation.

For the numerical simulations, the initial moisture contents (and the initial matric suctions) for the flow domain are established by allowing the vadose zone model to equilibrate with an infiltration rate representative of natural infiltration for tank farm location. For both WMAs C and S-SX, the representative infiltration rate was assumed to be 3.5 mm/yr (0.14 in./yr).

3.4.2.2 Recharge During Tank Farm Operations

Currently, each tank farm ground surface is covered with gravel to prevent vegetation growth and provide radiation shielding for site workers. Bare gravel surfaces, however, enhance net infiltration of meteoric water, compared to undisturbed naturally vegetated surfaces. Infiltration is further enhanced in tank farms by the effect of percolating water being diverted by the impermeable, sloping surface of the tank domes. This umbrella effect is created by the 75-ft (23-m) inside diameter of buried tank domes. Water, shed from the tank domes, then flows down the tank walls into the underlying sediments. Sediments adjacent to the tanks, while remaining unsaturated, can attain elevated moisture contents (Kline and Khaleel 1995). Enhanced infiltration from a gravel-covered tank dome can provide potential for faster transport of contaminants to the water table. Although site-specific infiltration data are being collected in BX, S, and T tank farms, insufficient data are available for site-specific estimates of natural infiltration at each farm.

For purposes of this SST PA, a reference case infiltration estimate of 100 mm/yr (3.93 in./yr) is used prior to barrier emplacement (Table 3-2). Data from experimental sites such as the Field Lysimeter Test Facility and the prototype Hanford barrier, both in the 200 Areas, suggest that recharge through gravels can vary from 15 to 70% of precipitation, with the lower amount occurring under vegetated conditions (Gee et al. 1996; Fayer and Walters 1995; Fayer et al. 1996).

3.4.2.3 Recharge beneath a Functioning Surface Barrier

Current plans are to use a Modified RCRA Subtitle C Barrier over the surface of each WMA as part of its closure (Section 1.7.3). The current pre-conceptual design for the Modified RCRA Subtitle C Barrier is based on DOE-RL (1996) and includes cover vegetation, silt loam layer(s), gravel layer(s), and an asphalt layer over the grading fill layer. The silt loam layer provides for moisture storage and allows evapotranspiration to occur before deep percolation can occur. The silt loam, along with the underlying gravel layer, forms a capillary break and impedes moisture flow across the interface. The design also includes an asphalt layer that provides biointrusion control and hinders human intrusion.

Extensive laboratory and modeling work and limited field testing of surface barriers have been performed; results are summarized in Fayer and Szecsody (2004). Lysimeter testing has been performed for different surface barrier concepts including a Modified RCRA Subtitle C Barrier with silt loam layers having depths between 1 and 2 m. Lysimeter data from the prototype Hanford barrier (Wing and Gee 1994) have also been collected and analyzed. Finally, modeling has been performed to address potential climate change impacts and no vegetation impacts on surface barrier performance.

The lysimeter drainage data that has been collected since 1989 suggests that the recharge rate beneath surface barriers having at least 1 m of silt loam is zero under ambient precipitation conditions. Most of these lysimeters did not contain an asphalt layer. Simulation results reported in Fayer and Szecsody (2004) investigated the sensitivity of the lysimeter data to climate change, silt loam hydraulic properties, vegetation changes, erosion, and dune formation above the surface barrier. Results indicated that the performance of these surface barriers was robust in that the estimated recharge rates remained below 0.1 mm/yr. For the cases investigated, only in the case of dune formation and no vegetation on the surface barrier were the simulated recharge rates above 0.1 mm/yr.

Based upon a review of the results, Fayer and Szecsody (2004) recommend an expected recharge performance for surface barrier with at least a 1 m of silt loam above a gravel layer to be on the order of 0.1 mm/yr for the life of the barrier. This estimate did not take any credit for the asphalt layer that is currently part of the Modified RCRA Subtitle C Barrier design.

The final design for the surface barrier has not been developed; however, based on the extensive testing reported in Fayer and Szecsody (2004), surface barriers that will limit recharge rates are achievable. For PA simulations involving tank farms with a functioning surface barrier, a reference case recharge rate of 0.5 mm/yr (0.02 in./yr) is assumed.

3.4.2.4 Recharge beneath a Degraded Surface Barrier

For a degraded surface barrier, a range of potential recharge rates can be envisioned. Fayer and Szecsody (2004) investigated the possibility of the most likely natural failure mechanisms (i.e., bioturbation of the silt loam layer, wind erosion, and accretion of wind blown sand). With appropriate design considerations, Fayer and Szecsody (2004) argue that the failure possibility of these natural systems is quite low, and the emplaced silt-loam soils will continue to perform for as long as they remain in place. Based on these arguments, Fayer and Szecsody (2004) concluded that the long-term effectiveness of the surface barrier would continue to limit recharge rates to less than 0.1 mm/yr for thousands of years.

Since the final design for the surface barrier has not been developed and it is difficult to defend the continued performance of a surface barrier for long periods of time, the SST PA has assumed the surface barriers will maintain the recharge rate at or below 0.5 mm/yr for 500 years. At the end of 500 years, the surface barrier performance is assumed to degrade to permit an infiltration rate of 1.0 mm/yr and maintain that infiltration rate for the remainder of the simulation for the reference case.

3.4.2.5 Reference Case Recharge Estimates

Table 3-2 summarizes the timeline estimates for barrier emplacement in tank farms and the corresponding reference case recharge estimates.

Table 3-2. Tank Farm Infiltration (Recharge) Estimates for Pre-Construction Period, Current Conditions, and Following Emplacement of Closure Barrier

Condition Simulated	Recharge Estimate mm/yr (in./yr)	Duration	Comment
Before construction of tank farms	3.5 (0.14) [a]	Until steady-state moisture conditions are achieved for the year, tank farm construction is completed	Vadose zone flow simulated at the recharge rate of 3.5 mm/yr to develop initial moisture conditions for subsequent simulations.
Current conditions	100 (3.93)	Construction completion year to year 2032	Recharge is assumed to increase from the pre-construction period estimate of 3.5 mm/yr to the current value of 100 mm/yr. During this period, the ground cover is gravel with no vegetation. A Modified RCRA Subtitle C Barrier is assumed to be in place by year 2032.
Transition to conditions of restricted recharge due to Modified RCRA Subtitle C Barrier	0.5 (0.02)	Years 2032 to 2532	Recharge is assumed to decrease from a current estimate of 100 mm/yr to the barrier design value of 0.5 mm/yr. The barrier is assumed to function to its design estimate of 500 years.
Degraded barrier condition	1.0 (0.14)	Years 2532 to 12032	The barrier is degraded and recharge increases from 0.5 mm/yr to 1.0 mm/yr until the end of simulation at year 12032.

[a] Based on 8-year lysimeter data for graveled surface (Fayer et al. 1999).

3.4.3 Grouted Tanks and Ancillary Equipment

The grouted tanks and ancillary equipment form one of the engineered barriers within the WMA closure system. The corresponding values and assumptions for the performance of this barrier are the parameters that were chosen to represent the contaminant source term releases from each of the contaminant sources and assumptions associated with its role in diverting moisture from the contaminant sources. This section describes the data used to select parameters used in the numerical simulations for the contaminant release from different waste forms, and the assumptions made concerning the role of the engineered barrier in diverting moisture from the contaminant sources.

3.4.3.1 Contaminant Source Term Releases

3.4.3.1.1 Past Releases. The contaminant sources associated with past releases were assumed to be readily available for transport with the infiltrating moisture; any chemical retardation is modeled by the linear isotherm K_d model discussed in Section 3.2.2.4. The estimates for the inventories for each identified release event are described in Section 3.4.1.2 for WMA C and WMA S-SX for selected contaminants and provided in Appendix C for all contaminants and other WMAs.

The distribution for each past tank leak in each WMA in the 200 West Area was modeled as a 25-ft (7-m) diameter source at a depth of approximately 130 ft (39.6 m) bgs located between tanks S-102 and S-103 for WMA S-SX. Section 2.7.5.2 provides a detailed discussion on the contamination that has been characterized in WMA S-SX. Gamma readings in the southern part of the SX tank farm are significant to a depth of 40 m bgs. This depth has been assumed to represent the depth for all known tank leaks in the 200 West Area because no contamination has been measured at a greater depth in the vadose zone, except for readings from one borehole in WMA TX-TY where contamination was observed at a depth of 150 ft bgs (see discussions for each WMA in the "Vadose Zone Conditions" subsections in Sections 2.6 through 2.12).
By selecting a maximum depth for all past tank leaks, the timing of the BTCs for past leaks is shortened and the peak contaminant concentration in the groundwater is maximized.
The selection of a 25-ft diameter of the source term was based on numerical simulations performed for the S-SX FIR (Knepp 2002a); calculations were performed for differing horizontal distributions for the past leak contaminants for WMA S-SX. These calculations showed the contaminant concentrations in the groundwater to be relatively insensitive to the assumed horizontal distribution of the contaminants from past leaks in the vadose zone. Based on these results, a contaminant inventory footprint diameter of 25 ft (7 m) was assumed for past leaks.

The distribution for each past tank leak in each WMA in the 200 East Area was modeled as a 25-ft (7-m) diameter source at a depth of 150 ft (45.7 m) bgs located between tanks C-112 and C-109 for WMA C. The depth of 150 ft (45.7 m) was assumed for each leak. The "Vadose Zone Conditions" subsections in Sections 2.10 through 2.12 summarize the current understanding based on characterization of vadose zone contamination for WMAs C, B-BX-BY, and A-AX. The deepest indications for vadose zone contamination are seen at 70 m bgs for WMA B-BX-BY, and 26 m bgs for WMA A-AX; no significant past leak contamination was reported for WMA C.

3.4.3.1.2 Tank Residual Wastes. The anticipated situation at the end of the retrieval process is as follows: a non-uniform distribution of tank residual waste within the tank with potentially higher accumulations distributed on strengthening rings on the side walls and equipment left in the tanks, and at selected regions on the tank bottom that are difficult to reach (due to the retrieval process utilized on the tank). The retrieval process is assumed to be successful in meeting HFFACO volume requirements (Ecology et al. 1989). For the reference case, the SST PA assumes that the tank as well as MUST wastes has been stabilized by filling them with cementitious grout. This assumption is consistent with closure planning. Residual wastes in pipelines within a WMA are assumed to represent unstabilized wastes.

Residual Waste Volume. For this SST PA, the volume of waste left in the tanks is assumed to meet HFFACO requirements of 360 ft^3 (10.2 kL) for 100-Series tanks and 30 ft^3 (850 L) in 200-Series tanks (Ecology et al. 1989). Given the dimensions of the tanks and the above assumptions, the average thickness of the residual waste in a 100-Series tank would be 1.0 in. (2.54 cm), while that for the 200-Series tanks would be 1.4 in. (3.6 cm).

Residual Waste Release Model. A diffusion-dominated release model was used to model release of contaminants from tank residual wastes for the reference case. The key parameters in this model are the effective diffusion coefficient for each contaminant in the waste and the mixing thickness. For the reference case, a diffusion coefficient of 1×10^{-9} cm^2/sec was used based on the Savannah River Site work on Hanford Site tank wastes (Harbour et al. 2004).

In Equation 3.6, the surface area to volume ratio, A_t/V_t, can be interpreted as the mixing length for the diffusional release from the tank bottom. The mixing length value used in the diffusion-dominated release model, however, is much larger than the uniform thickness of the residual wastes in the tanks. The mixing length accounts for the heterogeneous distribution of residual wastes within a tank, as well as the thick concrete structure for the diffusional release of contaminants from the tank. First, the residual wastes are not expected to be homogeneously distributed to a uniform thickness of 1 in. Following pouring of grout in the tank, the release from the tank occurs over a time period that exceeds thousands of years. During this long time period, contaminants within the residual wastes are transported by downward diffusion into the underlying concrete structure as well as by upward diffusion into "clean" grout.

For the concrete structure, Goetz (2003) states for the 100-Series tanks:

> "Each SST shell has an approximate 1-ft thick concrete base slab, dome and cylindrical wall that rests on a circular footing integral with the tank and base slab."

The upward diffusive length (L) is estimated to be about 18 in. Thus, for release calculations, the residual waste, prior to release, resides within the tank bottom thickness of 18 in. This plus the "average" 12-in. thickness plus other tank structural components (such as 2 in. of grout within the tank, 3/8 in. of mastic material [an asphalt liner], and 3/8-in. steel liner) accounts for the 0.825 m of mixing length used in this SST PA.

Other process-based release models are described later as part of sensitivity analyses (Section 3.5.4.3).

The numerical model calculation for the contaminant release from the tank residuals assumed a unit curie (Ci) source uniformly distributed at the bottom of each tank with the diffusional release starting on January 1, 2032.

3.4.3.1.3 Residual Ancillary Equipment Wastes. As defined in Section 2.5, other residual wastes include ancillary equipment such as pipelines and MUSTs within each WMA. For the reference case simulations, the residual waste inventory from pipelines was assumed to be homogeneously distributed. The release from pipeline residuals was modeled as a uniform inventory distributed over an approximately 7-m (approximately 25-ft) diameter at approximately 9 m (approximately 30 ft) bgs with the start of release occurring on January 1, 2000. The pipeline residual waste is assumed to be unstabilized and readily available for transport with the infiltrating water. The release from residual wastes in MUSTs is treated similar to release of residual waste from a tank, with the start of release occurring on January 1, 2032.

3.4.4 Vadose Zone

3.4.4.1 Moisture Flow and Contaminant Transport Data

With the exception of the surface barrier (Section 3.4.2), the operational period between years 2000 and 2032, and the consideration for the concrete structure in the contaminant release model for tank residual wastes (Section 3.4.3.1.2), facility structures are basically ignored in the moisture flow and contaminant transport simulations. The effect of the surface barrier is accounted for by changing the infiltration rate (Section 3.4.2.3). Other facility structures (e.g., the structural integrity of tank and ancillary equipment) are assumed to disappear after year 2032. This assumption was made to simplify the simulations and was chosen because it results in higher than expected contaminant concentrations in the unconfined aquifer. Thus, during the near term, the tanks act like umbrellas diverting moisture flow. In later years, the tanks were assumed to be filled with grout for the reference case; the grout properties impact the moisture flow. These grout properties were assumed to remain unchanged over the simulation period of 10,000 years. Moisture flow and contaminant transport data are provided for the near field, the vadose zone, and the unconfined aquifer beneath the WMA. The near field is defined as the region above the bottom of the tanks as shown in Figures 3-3 and 3-4. The vadose zone begins just below the horizontal line corresponding to the bottom of the tanks in the models.

3.4.4.1.1 Near Field Flow and Transport Properties. The recharge into the near field is described in Section 3.4.2. For the reference case, the tank structure was assumed to function as an impermeable barrier. The contaminant release as a function of time from the grouted residual tank waste is introduced into the numerical calculation at the bottom of each tank. No chemical retardation of the release was assumed.

3.4.4.1.2 Vadose Zone Flow and Transport Parameters. Details on vadose zone flow and transport parameters are provided in various data packages (Khaleel et al. 2006a, 2006b). The following sections summarize the methods and data used for the reference case.

Moisture Retention and Unsaturated Hydraulic Conductivity. The moisture retention data are described using an empirical relationship (van Genuchten 1980):

$$\theta(h) = \theta_r + (\theta_s - \theta_r)\left\{1 + [\alpha\, h]^n\right\}^{-m} \qquad \textbf{Eq. 3.8}$$

where:

θ = volumetric moisture content [dimensionless]
h = matric potential or pressure head, which, for notational convenience, is considered as being positive (i.e., tension [cm])
θ_r = residual moisture content [dimensionless]
θ_s = saturated moisture content [dimensionless]
α = a fitting parameter (cm^{-1})
n = a fitting parameter [dimensionless]
m = 1 - 1/n.

Combining the van Genuchten model with Mualem's (1976) model for unsaturated conductivity:

$$K(h) = \frac{K_s\left\{1 - (\alpha\, h)^{mn}\left[1 + (\alpha\, h)^n\right]^{-m}\right\}^2}{\left[1 + (\alpha\, h)^n\right]^{m\ell}} \qquad \textbf{Eq. 3.9}$$

where:

$K(h)$ = unsaturated hydraulic conductivity [cm/s]
K_s = saturated hydraulic conductivity [cm/s]
ℓ = pore-connectivity parameter [dimensionless], estimated by Mualem to be about 0.5 for many soils.

It is well recognized that the estimated unsaturated conductivities, based on saturated conductivity and the van Genuchten retention model, can differ by up to several orders of magnitude with measured conductivities at the dry end (e.g., Khaleel et al. 1995). A simultaneous fit of both laboratory-measured moisture retention and unsaturated conductivity data was used in this work, and all five unknown parameters θ_r, θ_s, α, n, and K_s, with m=1-1/n (van Genuchten 1980) were fitted to the data via a code named RETention Curve (RETC) (van Genuchten et al. 1991). Thus, in order to obtain a better agreement with experimental data for the region of interest (i.e., relatively dry moisture regime), K_s is treated as a fitted parameter during the curve-fitting process. The pore size distribution factor, ℓ (Mualem 1976) was kept fixed at 0.5 during the simultaneous fitting.

Table 3-3 lists the composite-fitted van Genuchten-Mualem (van Genuchten 1980; van Genuchten et al. 1991) parameters for various strata at WMAs C and S-SX. Estimates for the equivalent horizontal and vertical hydraulic conductivities are presented in the following section.

Table 3-3. Composite van Genuchten-Mualem Parameters for Various Strata at Waste Management Areas C and S-SX [a]

Strata	Number of Samples	θ_s	θ_r	α 1/cm	n	ℓ	Fitted K_s cm/s
Backfill	10	0.1380	0.0100	0.0210	1.374	0.5	5.60E-4
Sand H2	12	0.3819	0.0443	0.0117	1.6162	0.5	9.88E-5
Gravelly Sand H3	8	0.2688	0.0151	0.0197	1.4194	0.5	5.15E-4
Gravelly Sand H1	11	0.2126	0.0032	0.0141	1.3730	0.5	2.62E-4
Plio-Pleistocene	4	0.4349	0.0665	0.0085	1.8512	0.5	2.40E-4
Plio-Pleistocene/ Ringold Sandy Gravel	10	0.1380	0.0100	0.0210	1.374	0.5	5.60E-4

[a] Khaleel et al. (2006a; 2006b)

Moisture-Dependent Anisotropy. Vadose zone moisture-dependent anisotropy is used to account for the extensive lateral migration that is well documented for 200 East and West Areas sediments (Khaleel et al. 2006a; Ye et al. 2005; Yeh et al. 2005). For saturated media, an averaging of the heterogeneities in geologic media at a smaller scale leads to an effective hydraulic conductivity value at the larger (macroscopic) scale, with the lateral hydraulic conductivity being much larger than the vertical conductivity. For unsaturated media, theoretical and experimental analyses of field-scale unsaturated flow indicate that for stratified sediments such as those in the 200 Areas, the effective hydraulic conductivity tensor is anisotropic with a tension-dependent (or moisture-dependent) anisotropy. The anisotropy ratio of horizontal hydraulic conductivity to vertical hydraulic conductivity increases with increasing tension or decreasing moisture content. Because the soil hydraulic properties are based on small-scale laboratory measurements, upscaling methods are used to apply them to the large-scale, heterogeneous vadose zone (Khaleel et al. 2002). Tension-dependent anisotropy provides a framework for upscaling small-scale measurements to the effective (upscaled) properties for the large-scale vadose zone.

A stochastic model (Polmann 1990) was used to evaluate tension-dependent anisotropy for sediments at each WMA. Note that Polmann parameters (Table 3-4) were only used to assign anisotropy ratios for various strata within the vadose zone and are described by the following equation (Equation 3.10):

$$
\begin{aligned}
&< LnK > = < LnK_s > - A < h > - \sigma^2_{LnKs}\lambda[p - p^2 < h > - \zeta^2 < h >]/(1 + A\lambda) \\
&\sigma^2_{LnK} = \sigma^2_{LnKs}[(1 - p < h >)^2 + \zeta^2 < h >^2]/(1 + A\lambda) \\
&K_h^{eq} = \exp[< LnK > + (\sigma^2_{LnK}/2)] \\
&K_v^{eq} = \exp[< LnK > - (\sigma^2_{LnK}/2)]
\end{aligned}
\qquad \textbf{Eq. 3.10}
$$

where:

σ^2_{LnK} = variance of log unsaturated conductivity (which depends on mean tension)

$<h>$ = mean tension (positive) = $|\psi|$

ψ = matric potential (negative)

σ^2_{LnKs} = variance of lnK_s

$<LnK_s>$ = mean of lnK_s

p = slope of the β versus lnK_s regression line, where β is the slope of the unsaturated conductivity curve and approximated locally based on the Gardner's (1958) exponential model

ζ = $\sigma_\delta/\sigma_{lnKs}$

σ_δ = standard deviation of the residuals in the β versus lnK_s regression

A = mean slope, β, for lnK vs. h

λ = vertical correlation lengths for lnK_s (assumed to be same as that of β)

K_h^{eq} = equivalent unsaturated horizontal conductivity

K_v^{eq} = equivalent unsaturated vertical conductivity.

Table 3-4. Macroscopic Anisotropy Parameters for Various Strata at Waste Management Areas C and S-SX Based on Polmann (1990) Model [a]

Strata	Number of Samples	$<LnK_s>$	$\sigma^2_{LnK_s}$	p	ζ	λ cm	A
Backfill	10	-15.76	3.56	-1.1E-4	1.84E-4	30	0.00371
Sandy H2	12	-14.59	1.50	-7.2E-4	6.55E-4	50	0.00620
Gravelly Sand H3	8	-15.30	1.83	-5.6E-4	5.16E-4	50	0.00415
Gravelly Sand H1	11	-14.85	1.94	-2.6E-4	2.50E-4	30	0.00368
Plio-Pleistocene	4	-10.43	1.01	2.4E-3	9.34E-4	50	0.0104
Plio-Pleistocene/ Ringold Sandy Gravel	10	-15.76	3.56	-1.1E-4	1.84E-4	30	0.00371

[a] Khaleel et al. (2006a; 2006b)

Effective Transport Parameters. The reference case effective transport parameter (bulk density, diffusivity, and dispersivity) estimates are presented in this section. Because of natural variability, the transport parameters are all spatially variable. The purpose is similar to the flow parameters, to evaluate the effect of such variability on the large-scale transport process.

Bulk Density Estimates. Bulk density (ρ_b) estimates are needed to calculate retardation factors for different species. The average ρ_b, E[ρ_b] (Table 3-5) estimates for various strata are based on Khaleel et al. (2006a, 2006b).

Table 3-5. Effective Bulk Density (g/cm^3) Estimates at Waste Management Areas C and S-SX

Strata/Material Type	$E[\rho_b]$
Backfill	2.13
Sandy H2	1.76
Gravelly sand H3	1.94
Gravelly sand H1	2.07
Plio-Pleistocene silty sand	1.65
Plio-Pleistocene/Ringold gravels	2.13

[a] Khaleel et al. (2006a; 2006b)

Diffusivity. It was assumed that the effective, large-scale diffusion coefficients for all strata at a WMA are a function of volumetric moisture content, θ, and can be expressed using the Millington-Quirk (1961) empirical relation:

$$D_e(\theta) = D_0 \frac{\theta^{10/3}}{\theta_s^2} \qquad \textbf{Eq. 3.11}$$

where:
$D_e(\theta)$ = effective diffusion coefficient of an ionic species as a function of moisture content
D_0 = molecular diffusion coefficient for the same species in free water.

The molecular diffusion coefficient for all species in free water was assumed to be 2.5×10^{-5} cm^2/sec (Kincaid et al. 1995).

Macrodispersivity Estimates for Nonreactive Species. The Gelhar and Axness equation (Gelhar 1993) is used to estimate asymptotic values of macrodispersivity. To account for the effects of unsaturated flow, a modified version is used:

$$A_L(< h >) = \sigma_{LnK}^2 \lambda \qquad \textbf{Eq. 3.12}$$

where the longitudinal macrodispersivity depends on the mean tension $< h >$.

To apply Equation 3.12, an estimate of the vertical correlation scale for unsaturated conductivity is needed. A correlation length of the order of about 50 cm was obtained for saturated hydraulic conductivity for sediments near the C tank farm (Khaleel et al. 2006a). For unsaturated conditions, an increase in the variance of log unsaturated conductivity is expected to be compensated in part by a decrease in the correlation scale of log unsaturated conductivity. A correlation length of 30 cm is assumed for log unsaturated conductivity for all strata. Table 3-6 provides the log unsaturated conductivity variances and the estimated longitudinal (A_L) and transverse (A_T) macrodispersivities for various strata. The transverse dispersivities are estimated as one tenth of the longitudinal values (Gelhar et al. 1992).

Table 3-6. Nonreactive Macrodispersivity Estimates for Various Strata at Waste Management Areas C and S-SX

Strata	σ^2_{LnK}	Correlation length, λ cm	A_L cm	A_T cm
Backfill	4.54	30	~150	15
Sandy H2	4.60	30	~150	15
Gravelly Sand H3	4.95	30	~150	15
Gravelly Sand H1	3.19	30	~100	10
Plio-Pleistocene Sandy Silt	0.92	30	~50	5
Plio-Pleistocene/Ringold Sandy Gravel	4.54	30	~150	15

Heterogeneous Sorption Enhanced Macrodispersivities for the Reactive Species. The net effect of sorption is to retard the velocity at which the contaminant migrates through the porous media. Because sorption for specific contaminants may be a function of soil properties, as the soil properties experience spatial variability, the sorption also varies (Gelhar 1993; Talbott and Gelhar 1994). Stochastic analysis results for macrodispersivity enhancement for various strata in WMA C and S-SX are given in the modeling data package reports (Khaleel et al. 2006a, 2006b).

3.4.4.1.3 Contaminant Distribution Coefficients. Table 3-7 lists the contaminant distribution coefficient (K_d) bins used in this SST PA for various contaminants. Contaminant K_d values are adapted from:

- Section 4.3 of *Vadose Zone Hydrogeology Data Package for the 2004 Composite Analysis* (Last et al. 2004a)
- *Hanford Contaminant Distribution Coefficient Database and Users Guide* (Cantrell et al. 2002, 2003)
- *Soil Screening Guidance: Technical Background Document* (EPA 2000a)
- *Selection of K_d Values for INTEC Groundwater Modeling* (Jenkins 2001)
- *Engineering Design File – Fate and Transport Modeling Results and Summary Report* (INEEL 2004)
- *A Practical Guide to Groundwater and Solute Transport Modeling* (Spitz and Moreno 1996)
- *Risk Assessment Information System* (ORNL 2005).

A number of K_d bins are used to represent the range of sediment-contaminant chemical interaction for the variety of contaminants in various WMAs (Section 3.2.2.4.7).

Table 3-7. Contaminant Distribution Coefficients (mL/g) for Analytes that are Part of the Single-Shell Tank System Inventory (2 pages)

K_d = 0 mL/g			
Tritium	Tantalum	Oxalate	Neodymium
Carbon-14	Tellurium	Phosphate	N-nitroso-di-n-propylamine
Technetium-99	Tungsten	Benzyl alcohol	n-Nitrosomorpholine
Bismuth	Yttrium	Chloroethene	Phenol
Cerium	Acetate	Cyclohexanone	Pyridine
Chloride	Ammonia	Dichloromethane	Tetrahydrofuran
Chromium	Ammonium	Diethyl ether	Vanadium pentoxide
Praseodymium	Fluoride	Ethyl Acetate	2-Butanone(MEK)
Rhodium	Formate	Glycolate	2-Chlorophenol
Rubidium	Hydroxide	Isobutanol	2-Ethoxyethanol
Sodium	Nitrate	Lanthanum	2-Nitropropane
Sulfate	Nitrite	n-Butyl alcohol	2-Propanone
K_d = 0.02 mL/g			
Benzene	Nitrobenzene	1, 1, 1-Trichloroethane	1, 2-Dichloroethane
Carbon disulfide	Toluene	1, 1, 2, 2-Tetrachloroethane	4-Methyl-2-pentanone (MIBK)
Carbon tetrachloride	trans-1, 3-dichloropropene	1, 1, 2-Trichloroethane	
Chloroform	Trichlorofluoromethane	1, 1, 2-Trichloroethylene	
Ethylbenzene	Xylenes	1, 1-Dichloroethene	
K_d = 0.1 mL/g			
Cobalt-60	Cresylic acid	o-Nitrophenol	2, 4, 6-Trichlorophenol
Cobalt	m-Cresol	o-Xylene	2, 4-Dinitrotoluene
Chlorobenzene	o-Dichlorobenzene	Tetrachloroethylene	2-Methylphenol
K_d = 0.2 mL/g			
Iodine-129	Pentachlorophenol	1, 4-Dichlorobenzene	
p-Chloro-m-cresol	1, 1, 2-Trichloro- 1, 2, 2-trifluoroethane	2, 4, 5-Trichlorophenol	
K_d = 0.6 mL/g			
Uranium-232	Uranium-235 + D	Uranium	1, 2, 4-Trichlorobenzene
Uranium-233	Uranium-236	Naphthalene	
Uranium-234	Uranium-238 + D	Tributyl phosphate	

Table 3-7. Contaminant Distribution Coefficients (mL/g) for Analytes that are Part of the Single-Shell Tank System Inventory (2 pages)

K_d = 1.0 mL/g			
Ruthenium-106	Europium-152	Aluminum	Ruthenium
Cadmium-113m	Europium-154	Antimony	Thorium
Antimony-125	Europium-155	Cadmium	Tin
Tin-126	Radium-226 + D	Europium	Acenaphthene
Samarium-151	Radium-228 + D	Manganese	Ruthenium
K_d = 2.0 mL/g			
Selenium-79	Neptunium-237 + D	Plutonium-241 + D	Curium-244
Thorium-228 + D	Plutonium-238	Curium-242	Silver
Thorium-229 + D	Plutonium-239	Plutonium-242	Boron
Thorium-230	Plutonium-240	Americium-243 + D	Di-n-butylphthalate
Thorium-232	Americium-241	Curium-243	
K_d = 5.0 mL/g			
Actinium-227 + D	Zirconium-93	Mercury	Zirconium
Cesium-134	Arsenic	Molybdenum	Cyanide
Cesium-137 + daughters	Barium	Nickel	Aroclor-1254
Molybdenum-93	Beryllium	Selenium	Butylbenzylphthalate
Nickel-59	Calcium	Silicon	Di-n-octylphthalate
Nickel-63	Copper	Strontium	Fluoranthene
Niobium-93m	Iron	Thallium	Hexachlorobutadiene
Niobium-94	Lead	Titanium	Hexachloroethane
Protactinium-231	Lithium	Vanadium	Pyrene
Strontium-90 + D	Magnesium	Zinc	

Contaminant K_d values for many of the organic chemicals were estimated using the organic carbon partition coefficients (K_{oc}) method described in *Understanding Variation in Partition Coefficient K_d Values* (EPA 1999), and an estimated fractional organic carbon content for Hanford Site sediments of 0.03% (Cantrell et al. 2003). Contaminants for which inventory estimates exist and that do not have an assigned K_d value are assumed to have a $K_d = 0$. A soil material description of low organic, low salt, with a near-neutral pH is assumed. Appendix D contains a complete listing of contaminants and K_d bins. A brief discussion of each major contaminant follows.

Uranium. Uranium mobility is one of the most difficult features of the analysis to estimate. Significant work on uranium speciation and mobility is documented in Knepp (2002b) and Myers (2005). Last et al. (2004b) recommends the best estimate K_d value for uranium selected for most Hanford Site impact zones and source categories is 0.8 mL/g, with a range of 0.2 to 4. A reference case of 0.6 mL/g is selected for use in the reference case to represent a balance between possible conditions associated with tank wastes near the source and within the much larger vadose zone away from the source.

Technetium-99 (as Pertechnetate). The best estimates for the K_d values of pertechnetate are zero. The ranges were taken to be from zero to 0.1 mL/g for all source and impact zone categories (except gravel corrected). When comparing this range to values tabulated in Cantrell et al. (2002), the range may appear to be somewhat narrow; however, in most cases when higher K_d values were measured, the K_d values were not significantly greater than the standard deviation. As a result of this and the fact that it is known that pertechnetate is a very weak adsorbate, the narrow range for the K_d values was selected.

Iodine-129 (as Iodide). The value selected for the iodide K_d appropriate for most Hanford Site impact zones and source categories is 0.2 mL/g with a range of 0.1 to 2 mL/g. K_d values are assumed to be the same as those for groundwater.

Chromium. The geochemical behavior of chromium has been reviewed by Rai et al. (1988), Palmer and Wittbrodt (1991), Richard and Bourg (1991), and Palmer and Puls (1994). Ball and Nordstrom (1998) present a critical review of the thermodynamic properties for chromium metal and its aqueous ions, hydrolysis species, oxides, and hydroxides. A complete discussion on chromium geochemistry in the Hanford sediments can be found in Krupka et al. (2004).

Chromium(VI) as chromate (CrO_4^{2-}) is likely to be the dominant chromium species in the Hanford vadose zone and upper unconfined aquifer because its domain of predominance extends over a wide range of pH and Eh conditions that are appropriate for the vadose zone and upper unconfined aquifer. Chromium (III) complexes with dissolved ligands such as fluoride, ammonia, and cyanide (Baes and Mesmer 1976).

Limited studies infer that Cr(III), like other +3 cationic metals, is strongly and specifically absorbed by iron and manganese oxides present in soil (Korte et al. 1976). Cantrell et al. (2003) compiled in a single source, K_d values measured with Hanford sediment for radionuclides and CoCs that have potential human health effects in the vadose zone and groundwater at the Hanford Site. Cantrell et al. (2003, Table 10) list the K_d values determined for Cr(VI) for Hanford sediments; they found only a limited number of studies of Cr(VI) adsorption on Hanford sediments. The measured K_d values for Cr(VI) on Hanford sediments range from 0 to 1, with typical values being zero or close to zero, and based on these results, concluded that adsorption of Cr(VI) is very low to nonexistent under normal Hanford groundwater conditions unless conditions are acidic.

Nitrate and Nitrite. The behavior of nitrogen species, such as nitrate, in aqueous, soil, and geochemical systems has been discussed by Lindsay (1979), Lindsay et al. (1981), Stumm and Morgan (1981), Rai et al. (1987), Hem (1986), and others. A large number of studies have been completed related to the chemical and biological processes that transfer nitrogen between the atmosphere, lithosphere, hydrosphere, and biosphere. Nitrate (NO_3^-) is highly mobile and does not sorb or precipitate in sediment systems.

Nitrate is one of the most widespread contaminants associated with past Hanford Site operations. Nitrate does not readily adsorb on minerals under near-neutral and slightly alkaline pH conditions common in sediment systems, and is typically not included in most databases of K_d. Nitrate and nitrite (NO_2^-) are typically assigned K_d values of 0 mL/g. Cantrell et al. (2003) identified only one study in which nitrate adsorption was measured using Hanford sediments. The limited number of K_d values determined for nitrate from this study are listed in Table 12 of Cantrell et al. (2003). Based on these measurements, Cantrell et al. (2003) concluded that within experimental error, nitrate adsorption under Hanford Site relevant conditions is essentially zero (i.e., $K_d = 0$).

Table 3-8 summarizes the contaminant distribution coefficients used for the reference case and the measured ranges in these values for non-impacted soils.

Table 3-8. Contaminant Distribution Coefficients (mL/g) for Non-Impacted Soils [a, b]

Contaminant	Reference Case	Minimum	Maximum
Uranium	0.6	0.2	4
Iodine-129	0.2	0.1	2
Technetium-99	0	0	0.1
Nitrite	0	0	0
Nitrate	0	0	0
Chromium	0	0	0.3

[a] Refers to far-field modeling domain with ambient conditions, and unaffected by past releases and tank leak chemistry.

[b] Last et al. (2004b)

3.4.4.2 Use of Waste Management Areas C and S-SX Breakthrough Curves for Other Waste Management Areas

Detailed contaminant transport calculations using STOMP have only been performed for WMA C and WMA S-SX. Detailed calculations are planned for the other WMAs and will be incorporated into future revisions of the SST PA. To provide an estimate of the human health impacts anticipated from other WMAs that have not been explicitly modeled, the modeling results from WMA C calculations have been used to scale the estimated impacts for other WMAs in the 200 East Area. Similarly, the modeling results from the WMA S-SX calculations have been used to scale the estimated impacts from other WMAs in the 200 West Area. Section 3.2.2.4.8 provides a description of this approach and discusses the reasons why such an approach can provide reasonable estimates for the impacts for other WMAs.

3.4.5 Unconfined Aquifer

For the integrated, saturated-unsaturated, two-dimensional, cross-sectional model up to the WMA fenceline, the flow parameters needed for unconfined aquifer calculations are saturated hydraulic conductivity, effective porosity, hydraulic gradient, and depth to water table. These parameters for WMAs C and S-SX are given in Table 3-9. Estimates of hydraulic properties shown in Table 3-9 are taken from the work of Wurstner et al. (1995) that was used to develop the Hanford Site groundwater model. Hydraulic gradients are based on the Hanford Site-wide groundwater model (Wurstner et al. 1995; Cole et al. 2001b) estimates of post-Hanford conditions.

Table 3-9. Reference Case Unconfined Aquifer Properties for Waste Management Areas C and S-SX

Property	WMA C	WMA S-SX
Saturated hydraulic conductivity (meters/day)	3,000	25
Effective porosity (unitless)	0.25 (Hanford gravel)	0.10 (Ringold)
Hydraulic gradient (unitless)	1×10^{-5}	5×10^{-4}
Depth to water table (m)	79	78

3.4.6 Exposure Parameters

The reference case analysis exposure scenario for the groundwater pathway is industrial use. For comparative purposes, the all-pathway farmer scenario and residential scenario for ILCR are also considered (Chapter 1.0). The reference case exposure scenarios for the intruder pathway are acute exposure to a well driller and post-intruder rural pasture. Other intruder scenarios considered in this SST PA include the suburban garden and commercial farmer. These exposure scenarios are discussed in Chapter 1.0. The dose parameters are based on Rittmann (2004).

3.4.7 Reference Case

The reference case incorporates the most reasonable and representative information currently available. As more information concerning the waste form, closure design, and closure site locations are gathered, the definition of the reference case is expected to evolve.

The details of the models and related data for the reference case are presented in this section. The major features of the reference case are as follows:

- The future land use of the 200 Areas is envisioned as being a protected area (i.e., with no artificial recharge from irrigated farming).
- The source terms for the risk assessment consist of three separate sources following closure: 1) past releases, 2) residual waste in the tanks following closure, and 3) residual waste in the ancillary equipment following closure (Section 3.4.3.1).
- The release mechanism for past releases is modeled as being transported with natural recharge (Section 3.4.3.1.1).
- The residual tank waste follows the HFFACO goal of 360 ft^3 (10.2 kL) for 100-Series tanks and 30 ft^2 (850 L) for 200-Series tanks or 1% residual waste volume for both 100- and 200-Series tanks (Ecology et al. 1989) (Section 3.4.3.1.2).
- The release mechanism for tank residual wastes is modeled as being diffusion-dominated with a diffusion coefficient of 1×10^{-9} cm^2/sec (Section 3.4.3.1.2).

- The recharge rates before construction of the tank farms, for the tank farm operation period up to year 2032, for the non-degraded barrier (years 2032 to 2532), and for the degraded barrier (years 2532 to 12032) are assumed to be the same for all WMAs in the 200 West Area (3.5, 100, 0.5, and 1 mm/yr, respectively). The recharge rates for the different time periods are assumed to be the same for all WMAs in the 200 East Area (3.5, 100, 0.5, and 1 mm/yr, respectively) (Section 3.4.2).
- The vadose zone flow and transport properties developed from field and laboratory measurements conducted in the 200 Areas were used in the vadose zone modeling to calculate BTCs for contaminants in the groundwater at the WMA C fenceline (Section 3.4.4.1.2).
- The vadose zone flow and transport properties developed from field and laboratory measurements conducted in the 200 Areas were used in the vadose zone modeling to calculate BTCs for contaminants in the groundwater at the WMA S-SX fenceline (Section 3.4.4.1.2).
- The unconfined flow and transport properties for WMAs in the 200 East Area are based on those for WMA C, whereas the properties for WMAs in the 200 West Area are based on those for WMA S-SX (Section 3.4.5).
- The BTCs developed for WMA C for each contaminant source term were used to estimate the BTCs for other WMAs in the 200 East Area (WMAs B-BX-BY and A-AX) (Section 3.4.4.2).
- The BTCs developed for WMA S-SX for each contaminant source term were used to estimate the BTCs for other WMAs in the 200 West Area (WMAs T, TX-TY, and U) (Section 3.4.4.2).

The following sections provide a rationale for the selection of the sensitivity analysis case values and assumptions.

3.5 SENSITIVITY CASES

3.5.1 Overview

This section describes the sensitivity analyses and the associated parameter estimates used to evaluate environmental contamination and human health impacts associated with the tank closure system. The sensitivity analysis provides information (Section 4.11 and Chapter 5.0) that addresses the following issues:

- How well can the performance of the closure system be estimated?
- How important are the "barriers" to the performance of the system?
- What is the value of additional information to reduce estimated uncertainty?

These issues arise largely because of inherent variability within the tank closure system. Heterogeneities in the natural system, long-term degradation of engineered barrier performance, and future human actions can affect future environmental contamination. Such variability generates uncertainty about real-time contaminant migration characteristics and limits the ability of the modeler to adequately portray system features and processes that affect future environmental contamination levels (e.g., predictive uncertainty). Because tank closure system variability cannot be completely resolved (also referred to as irreducible variability), a range of future environmental impacts are estimated that account for effects of this variability and provide a qualitative measure of uncertainty. To complete these estimates, a suite of sensitivity cases has been formulated in which variability was quantified as ranges of modeling input parameter values that envelope reference case values. The resulting set of changes in estimated environmental (e.g., groundwater) contamination levels in combination with reference case results provided ranges of plausible future contamination levels caused by tank closure system variability effects.

Of the three major pathways (i.e., groundwater, air, and intruder), the most complicated analysis involves the groundwater pathway. Contaminant migration through the subsurface is influenced by all major components of the closure system. Therefore, the greatest amount of potentially significant parameter variability is associated with this pathway. As with the reference case analysis (Section 3.4), the bulk of this section is devoted to this pathway. Section 3.5.2 is a description of the sensitivity analysis methodology; Sections 3.5.3 and 3.5.4 describe selected cases and associated parameters and parameter values.

Sensitivity analyses for the intruder pathway were focused on a range of human activities (e.g., scenarios) that cause variable levels of exposure to contaminants. Four exposure scenarios, one acute and three chronic, are described in Chapter 5.0, and parameter changes affecting the assumed amounts of exhumed waste and degree of mixing with soil are summarized in Section 3.5.5. No sensitivity analyses were conducted for the air pathway. The air pathway analysis is a bounding analysis that provides an implausible maximum environmental impact that satisfies the regulatory criteria (Section 3.5.6). Consequently, a sensitivity analysis was considered unnecessary.

Groundwater pathway sensitivity analyses provided a plausible range of future groundwater contamination levels derived from closure system variability effects.

Sources of variability include natural system heterogeneities, long-term engineered barrier performance degradation, and future human actions.

Intruder pathway sensitivity analyses estimated human heath effects for various human activity exposure scenarios and exposure pathways.

3.5.2 Sensitivity Analysis Methodology for the Groundwater Pathway

The methodology for the groundwater pathway uses a deterministic approach to calculate a plausible range of future groundwater contamination levels at the WMA fenceline that is generated by site-specific closure system variability and encompasses the reference case result. This approach is deterministic because potential groundwater contamination levels are determined for discrete parameter values that define ranges in parameter (e.g., minimum, reference case, and maximum values). Collectively, the suite of analytical results defines a finite range of plausible future groundwater contamination levels. Although there is a qualitative expectation that the actual groundwater contamination levels should tend toward the reference case estimate (e.g., the purpose of the reference case assumptions is to provide the best estimate of actual closure system conditions), the analyses do not assign a likelihood of occurrence to a particular outcome. These results contrast with those provided by a probabilistic/stochastic approach, in which a continuum of parameter values are often considered and their likelihood of occurrence is assumed. These assumptions are then propagated through the analysis to a set of realizations where the likelihood of a particular outcome occurring can be calculated.

The approach described herein was selected for the following reasons:

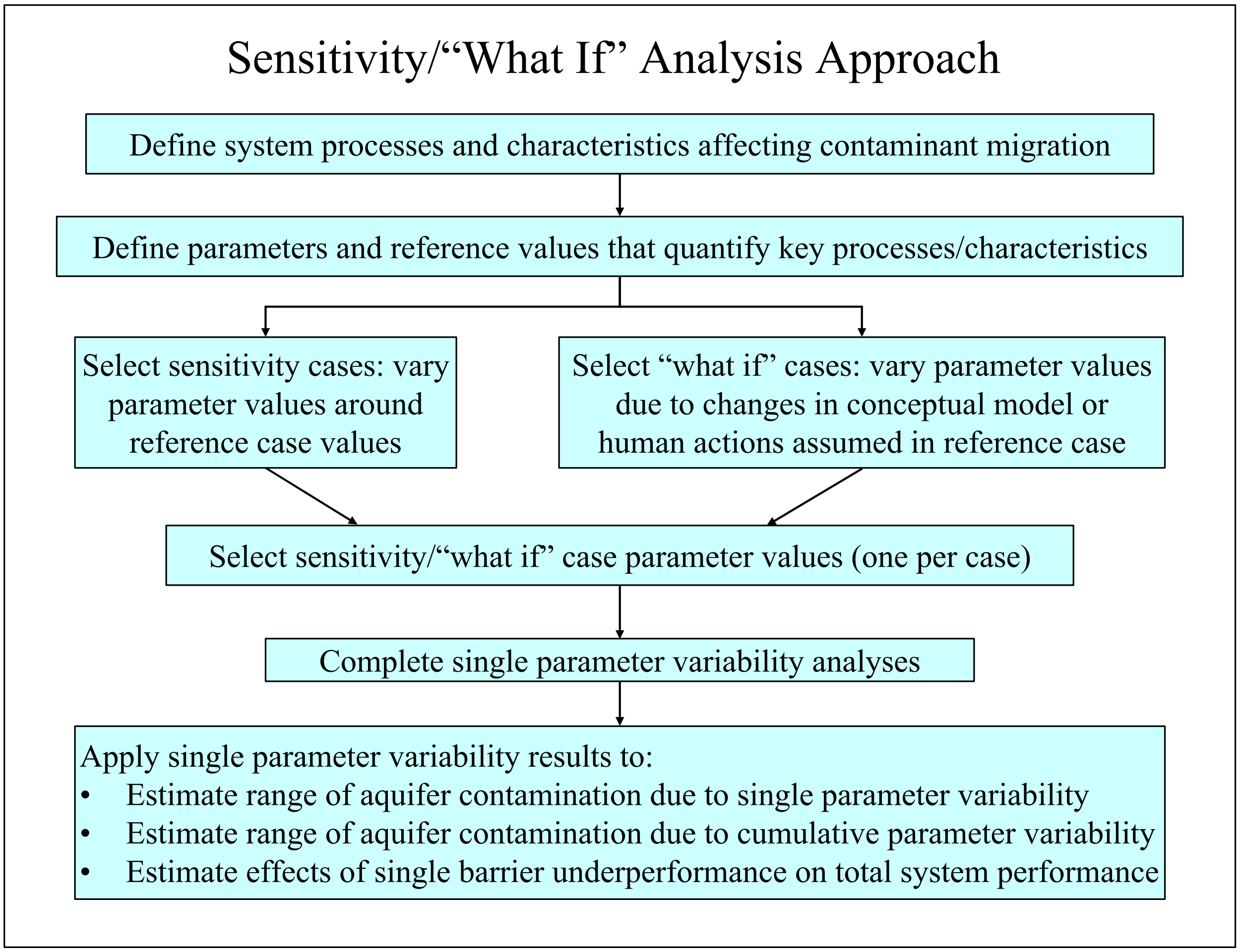
- Closure system performance is being compared with respect to numerous deterministic criteria (e.g., groundwater protection criteria and various human health effects). Comparison of deterministic criteria with deterministic analytical results provides a transparent indication of acceptable or unacceptable performance.
- The available database describes the major features and processes affecting contaminant transport in some detail (e.g., precipitation and infiltration rates, subsurface hydrogeologic characteristics, contaminant-specific geochemical behavior) and is amenable to the quantification of minimum and maximum values of critical parameters. The database, for the most part, is not considered adequate to assign probability distribution functions to various parameters.
- Manipulation of single and multiple changes in parameter values coupled with associated changes in future groundwater contamination levels provides insight into the relative importance of various features and processes affecting contaminant transport. These results also provide estimates of plausible variability in future groundwater contamination levels (e.g., uncertainty around the reference case outcome) and illustrate the estimated range of plausible outcomes due to irreducible system variability.
- This approach identifies additional important data needs. Importance is defined as data that are currently unavailable and are needed to better quantify a parameter value range that may generate relative large changes in projected groundwater contamination levels.

A flow chart of the sensitivity case methodology for the groundwater pathway is provided in Figure 3-13. In this methodology, sensitivity cases were derived to evaluate the effects of system variability, groundwater contamination analyses were completed for each case, and comparisons of sensitivity case results (estimated groundwater contamination levels at the WMA fenceline) to reference case results were made.

Initially, sensitivity cases were derived from the reference case modeling assumptions (Section 3.2-3.4) which defined the following:

- Site-specific closure system features and processes affecting contaminant migration
- Parameters that describe these features and processes
- Site-specific reference parameter values.

Figure 3-13. Modeling Approach for Evaluating Closure System Variability Effects on Estimated Future Groundwater Contamination Levels Relative to Reference Case Results



Critical features and processes included site-specific natural system characteristics such as low infiltration rates, a thick vadose zone, the current distribution of contaminants with the vadose zone, and engineered components including the surface barrier, and the grouted tank structure. Significant processes included unsaturated flow in the vadose zone, contaminant-specific geochemical reactivity with the subsurface sediments, and mixing of contaminated vadose zone water with clean unconfined aquifer water. Parameters describing these features and processes fell into three broad categories: recharge rates, waste characteristics (e.g., inventory and release mechanisms), and geohydrologic properties of the vadose zone and unconfined aquifer.

To proceed with the sensitivity analysis, two types of cases were distinguished, sensitivity cases and "what if" cases. The sensitivity cases (Section 3.5.3) assumed all primary reference case assumptions were unchanged and simply varied parameter values with respect to reference case values. In the "what if" cases (Section 3.5.4), alteration of postulated reference case assumptions affecting contaminant migration was assumed (e.g., different physical or chemical processes or human actions that altered system conditions). To represent these different assumptions, different parameter estimates were considered. In these analyses, only contaminant migration to the WMA fenceline and only migration from waste sources in WMAs C and S-SX (representing 200 East and West Area geohydrology, respectively) were considered. Also, sensitivity analysis results were only generated for constituents that reached the unconfined aquifer in reference case analyses.

Each sensitivity and "what if" case usually assumed a change in one parameter value and, in some cases, the substitution or addition of a parameter relative to the reference case. By grouping cases that considered value changes to the same parameter for specific contaminants present in specific waste sources, a value range for a given parameter was generally defined by at least three values, a reference case value, a minimum value, and a maximum value. For example, the reference case operational period recharge rate was 100 mm/yr and two sensitivity cases were generated that assumed operational period recharge rates of 40 and 140 mm/yr to define the minimum and maximum values, respectively (Section 3.5.3.1). Then, by completing a contaminant migration analysis that estimated the groundwater impacts for each sensitivity/"what if" case and associated parameter value change within each parameter group, a range of groundwater contamination levels was generated in response to parameter value change.

Changes in groundwater contamination levels at the WMA fenceline in response to parameter value changes were calculated and, for ease of comparison, expressed as ratios of the peak groundwater contamination levels from the sensitivity case to peak levels estimated in the reference case (Sections 4.11.1 through 4.11.4). These ratios, referred to as peak ratios, indicated the sensitivity of contaminant migration to variability of a particular parameter because each ratio was associated with specific single parameter value changes (e.g., single parameter variability). Estimated increases or decreases in groundwater contamination levels were indicated by peak ratio values greater or less than unity, respectively. Relatively larger or smaller ratios for given parameter ranges indicated greater sensitivity of groundwater contaminant levels to variability of the feature or process represented by the parameter. These ratios also indicated uncertainty around the reference case estimate with respect to variability of a particular parameter. That is, the plausible range of estimated groundwater contamination levels was constrained by the plausible range of site-specific parameter values determined from site-specific data.

The results of the single parameter variability analyses were then used for two additional applications, a cumulative variability analysis (Section 4.11.4.5) and a barrier underperformance analysis (Section 4.11.5). The calculation processes for these two applications are provided in their respective sections. The cumulative variability analysis estimated the effects of multiple and simultaneous parameter value changes on groundwater contamination levels for given contaminant/waste type combinations. Because the effects of variability in all significant parameters were considered at once, a cumulative uncertainty in peak groundwater contamination levels with respect to the reference case estimates was provided.

The barrier underperformance analysis estimated the effects of single or multiple barrier underperformances (i.e., the surface barrier, the grouted tank structure, and/or the vadose zone) on total system performance with respect to reference case assumptions. In these analyses, parameters describing barrier functions and associated sensitivity/"what if" case results were selected. Using peak ratios, underperformance factors were calculated to estimate the potential increases in groundwater contamination levels with respect to degradation of single or multiple barriers. Again, because estimated groundwater contamination increases were tied to plausible parameter value ranges, uncertainty in system performance was estimated.

3.5.3 Estimated Ranges for Selected Parameters in Groundwater Pathway Sensitivity Cases

Estimated ranges for recharge, contaminant release from residual waste and past releases, vadose zone parameters, contaminant distribution coefficients, and unconfined aquifer parameters are discussed in Sections 3.5.3.1 through 3.5.3.5.

3.5.3.1 Recharge

As discussed earlier, recharge is a major driver for contaminant transport from various waste forms to groundwater and to an eventual receptor. Recharge information is needed for four time periods associated with tank farm closure: 1) prior to construction of tank farms, 2) during tank farm operations, 3) the period when the emplaced surface barrier performs as designed, and 4) the period when the surface barrier performance is degraded. Although site-specific infiltration data are being collected in the BX, S, and T tank farms for the operation period, insufficient data are available for site-specific estimates of natural infiltration for a given WMA. Based on results of long-term lysimeter experiments, a reference value (100 mm/yr) is used in this assessment for the present-day, gravel-covered tank farms, before placement of the Modified RCRA Subtitle C Barrier. This value is based on 9 years of testing at the Field Lysimeter Test Facility for a surface treatment of "sandy gravel side slope" (Fayer and Szecsody 2004, p. A.20) and is rounded upward from the reported value of 99.8 mm/year (Fayer and Szecsody 2004, p. 7.8).

To examine the effect of expected ranges in this parameter, a lower value (40 mm/yr) and a higher value (140 mm/yr) were selected. The 140 mm/year is the result of an enhanced precipitation experiment on the "sandy gravel side slope" treatment. This treatment is referenced as "useful for characterizing deep drainage at the high-level waste tank farms at Hanford" (Fayer and Szecsody 2004, p. A.6). Enhanced precipitation represented three times the average precipitation. Approximately 140 mm of the precipitation was observed to have infiltrated.

The lower value is supported by the prototype barrier testing summary results from Wittreich et al. (2003). On the basis of 7 years of monitoring data, Wittreich et al. (2003) reported that the annual drainage rate of 21.5% of precipitation (37.8 mm/yr) through sparsely vegetated sandy gravel representing the "sandy gravel side slope" became recharge. Such a percentage (i.e., 21.5%) represents a lower value of approximately 40 mm/yr.

The surface barrier is an important engineered barrier for a WMA. Once it is emplaced, the surface barrier performance directly impacts the recharge into the WMA. The current pre-conceptual design of the Modified RCRA Subtitle C Barrier is based on DOE-RL (1996). The barrier is designed to provide containment and long-term hydrologic protection for a period of 500 years and consists of 8 layers. The design accounts for human and biointrusion control, and is based on a silt loam moisture storage unit and a capillary break. The thickness of the surface cover over the WMAs is assumed to be at least 5 m. The barrier is assumed to be thick enough to ensure that the waste resides at a depth of at least 5 m below the barrier surface.

Recent work (Fayer and Szecsody 2004) recommends an expected performance for such a barrier to be on the order of 0.1 mm/yr for the life of the barrier. The report also concludes that, with appropriate design considerations, the possibility of the most likely natural failure mechanisms (i.e., biointrusion of the silt loam layer, wind erosion, and accretion of wind blown sand) to occur is quite low, and that the emplaced silt-loam soils will continue to perform as designed. Fayer and Szecsody work has not been extended to address the range of conditions that might be encountered leading to a decline in performance of surface barriers placed on tank farms. For this reason, and due to the lack of any detailed surface barrier design for tank farm closure, the reference case infiltration rate is set at 0.5 mm/yr for the first 500 years of the simulation. This value is the performance requirement used for the design of the Hanford barrier. A range of 0.1 to 1.0 mm/yr is selected to address the expected range in this parameter over the 500-year design life for the barrier. The reference infiltration rate is five times above the recommended reference value (Fayer and Szecsody 2004). The upper range of 1.0 is simply 10 times the expected performance estimate of 0.1 mm/yr, while the lower estimate is the expected performance recommendation (Fayer and Szecsody 2004). Given that the source of construction materials are identical for the tank farm barriers and for the Integrated Disposal Facility PA assumed in Fayer and Szecsody's work, a barrier design value of 0.5 mm/yr is considered easily attainable and will likely be improved as the tank farm closure project advances.

At the end of 500 years, the surface barrier infiltration rate is assumed to degrade to 1.0 mm/yr and maintain that infiltration rate for the remainder of the simulation. A range of 0.5 mm/yr to 3.5 mm/yr for the 200 East and 200 West Area is further assumed. The upper ranges are based on soil types and infiltration characteristics of the native soils as reported in *Geographic and Operational Site Parameters List (GOSPL) for the 2004 Composite Analysis* (Last et al. 2004b). The lower range represents a case whereby the surface barrier remaining intact. Currently, only the assumed performance of the barrier is simulated.

3.5.3.2 Contaminant Release from Tank Residual Wastes and Past Releases

The reference case assumes that the rate at which the grouted tank residual waste is made available for transport to groundwater is through the process of diffusion (Section 3.4.3.1.2). No allowance is made for other potential barriers (such as the tank structure) that would likely delay release.

The DOE Office of River Protection has identified three distinct functions for tank fill (Lee 2004). Grout formulations are designated to correspond to each of these three different functions:

- Stabilizing grout to eliminate residual liquid and stabilize contaminants in the tank residues
- Structural grout to provide structural support
- Capping grout to provide an intruder barrier.

Leaching tests (ANSI/ANS 16.1) of stabilizing grout formulations on simulated Hanford Site tank wastes led to the following recommendation (Harbour et al. 2004, p. 2):

> "Calculations of the effective diffusivities using measured Tc-99 values resulted in D_e of 1.9 E-09 to 2.1 E-10 cm^2/sec. The range using plus/minus 2 sigma (standard deviation) values for the Tc-99 concentrations gave an overall range from 1.3 E-09 to 1.1 E-14 cm^2/sec. These data can be used in support of PA calculations."

For this SST PA, a minimum D_e of 1 E-14 cm^2/sec and a maximum D_e of 1 E-8 cm^2/sec are used.

Past Releases. The estimate of past tank leaks and spills from tank farms into the vadose zone has undergone significant investigation in the past few years. Releases within a tank farm are highly variable with the largest leaks found in SX and T tank farms (Field and Jones 2005). The association of past releases with the specific operational history for each WMA requires an estimate of an expected range of release be developed on a WMA-specific basis. Tables 3-10 and 3-11 present such an evaluation for WMAs C and S-SX, respectively. All estimates are provided in gallons. Inventories of radiological and nonradiological contaminants will be linearly scaled based on the contaminant concentration estimates provided for the reference case and listed in Corbin et al. (2005).

The range in the maximum and minimum release volume estimates are conditioned on the available information for each event. Well-documented tank leaks and spills generally have low variability, while smaller tank leaks and spills have larger variability.

Table 3-10. Estimates of Ranges for Past Releases Within Waste Management Area C [a]

Tank/Spill	Reference Case Leak Volume gal	Minimum Leak Volume gal	Maximum Leak Volume gal	Comments
C-101	1,000	500	5,000	Liquid losses from this tank are adequately explained by evaporative cooling.
C-105	1,000	500	1,000	Vadose zone contamination most likely came from pipe leaks.
C-110	2,000	500	2,000	—
C-111	5,500	500	5,500	Historical data show that the liquid level decreases are associated with evaporative cooling.
C-201	550	550	1,000	Liquid losses are likely due to evaporation.
C-202	450	450	1,000	Liquid losses are likely due to evaporation.
C-203	400	400	1,000	Liquid losses are likely due to evaporation.
C-204	350	350	1,000	Liquid losses are likely due to evaporation.

[a] Unplanned releases and ancillary equipment releases are provided in Table 3-1.

Table 3-11. Estimates of Ranges for Past Releases Within Waste Management Area S-SX [a]

Tank/Spill	Reference Case Leak Volume gal	Minimum Leak Volume gal	Maximum Leak Volume gal	Comments
S-104	24,000	19,000	29,000	Well-documented leak volume. Assumed range is 20% of reference case.
SX-104	6,000	500	6,000	Liquid losses are likely due to evaporation.
SX-107	15,000	5,000	20,000	—
SX-108	35,000	15,000	50,000	—
SX-109	2,000	1,000	5,000	—
SX-110	1,000	500	1,000	—
SX-111	500	500	2,000	—
SX-112	1,000	500	5,000	—
SX-113	15,000	12,000	18,000	Well-documented leak volume. Assumed range is 20% of reference case.
SX-114	500	500	1,000	—
SX-115	50,000	40,000	60,000	Well-documented leak volume. Assumed range is 20% of reference case.

[a] Unplanned releases and ancillary equipment releases are provided in Table 3-1.

3.5.3.3 Vadose Zone Parameters

Sensitivity analysis associated with unsaturated flow parameters has been conducted in the past (DOE-RL 1999; Mann et al. 2001). Historical work (e.g., Mann et al. 2001) indicates this feature of the system is generally of secondary importance to others, such as recharge rates and inventory estimates of past releases. For the purpose of this SST PA, to account for sensitivity of unsaturated flow parameters, the saturated hydraulic conductivity is used as a scaling factor; the parameter variability is assumed to have a range of ± one order of magnitude from the reference case.

As discussed earlier (Section 3.2.2.4.7), the effect of discrete features such as clastic dikes on vadose zone flow and transport were investigated in Knepp (2002a). Specific simulations were run to evaluate the effects of clastic dikes on the peak groundwater concentration at the WMA fenceline in the S-SX FIR (Knepp 2002a). As discussed in Section 3.2, simulated groundwater peak concentrations for long-lived mobile radionuclides were not significantly different for simulation cases with or without dikes. However, as discussed later, a case is simulated where a retrieval leak of 8,000 gal is assumed to occur for a 100-Series tank in the vicinity of a postulated clastic dike.

3.5.3.4 Contaminant Distribution Coefficients

Table 3-8 presents a summary of the reference case, and maximum and minimum ranges of contaminant K_d values for contaminants affecting risk through the groundwater pathway. These K_d values have been used for contaminant transport calculations through the vadose zone and through the unconfined aquifer to the WMA fenceline.

3.5.3.5 Unconfined Aquifer Parameters

Two distinct hydrologic regimes must be considered when examining the potential range of unconfined aquifer properties. Groundwater within the western portion of the Central Plateau is contained within the Ringold Formation, while groundwater within the eastern portion of the Central Plateau is generally found in the Hanford formation. The Ringold Formation is a semi-indurated, intercalated, coarse-grained fluvial sequence. For the Ringold Formation, variability in aquifer properties is generally low in comparison to the Hanford formation. The Hanford formation is predominately fluvial sand and gravel, often composed of open framework gravels having extremely high transmissivities.

Saturated media variables that impact modeling results include horizontal hydraulic conductivity, effective porosity, and hydraulic gradient. Hydraulic properties have been measured over the past 50 years and are documented in a number of reports (DOE 1988; Thorne and Newcomer 1992; and more recently, Wurstner et al. 1995). The aquifer saturated hydraulic conductivity in the vicinity of WMA S-SX is generally quite low, increases to the north, and finally decreases toward the northern edge of the 200 West Area. Hydraulic conductivities in the 200 East Area are generally much higher and are highly dependent on the presence of fine or coarse aquifer materials. Ranges of hydraulic properties shown in Tables 3-12 and 3-13 are based on Wurstner et al. (1995) and were used to develop the Hanford Site groundwater model. Much less information is available on the variability in porosity for either formation. Effective porosities shown in Tables 3-12 and 3-13 are the values generally used for the respective formations (Khaleel et al. 2000, 2001). Hydraulic gradients are based on the Hanford Site-wide groundwater model (Wurstner et al. 1995; Cole et al. 2001b) estimation of post-Hanford conditions. Tables 3-12 and 3-13 provide estimates for WMAs C and S-SX, respectively, of the recommended ranges for aquifer properties for use in the sensitivity analysis.

Table 3-12. Unconfined Aquifer Properties for Waste Management Area C

Property	Reference Case	Minimum	Maximum
Hydraulic conductivity (m/day)	3,000	2,000	4,000
Effective porosity (unitless)	0.25 (Hanford gravel)	NC	NC
Hydraulic gradient (unitless)	0.00001	NC	NC
Depth to water table (m)	79	NC	NC

NC = not considered

Table 3-13. Unconfined Aquifer Properties for Waste Management Area S-SX

Property	Reference Case	Minimum	Maximum
Hydraulic conductivity (m/day)	25	7.5	50
Effective porosity (unitless)	0.1 (Ringold)	NC	NC
Hydraulic gradient (unitless)	0.0005	NC	NC
Depth to water table (m)	78	NC	NC

NC = not considered

Table 3-14 provides a summary of the parameters and their ranges for the natural and engineered barriers and features for the closed WMA. The minimum and maximum parameter ranges were used in the sensitivity analysis.

Table 3-14. Groundwater Pathway – Summary of Reference Case Parameters and Expected Ranges

Natural and Engineered Barriers/Features	Feature/Process	Reference Case	Parameter Range	
			Minimum	Maximum
Surface cover	P1: Infiltration	An infiltration rate of 100 mm/yr for the reference case during tank farm operation up to year 2032.	40 mm/yr	140 mm/yr
	P2: Infiltration	An infiltration rate of 0.5 mm/yr for the reference case for the barrier from years 2032 to 2532.	0.1 mm/yr	1.0 mm/yr
	P3: Infiltration	An infiltration rate of 1.0 mm/yr for the reference case for the barrier from years 2532 to 12032.	0.5 mm/yr	3.5 mm/yr
Grouted tank structure	P4: Residual release – Diffusion coefficient	Diffusion-dominated release for residual tank wastes with a diffusion coefficient of 1×10^{-9} cm^2/sec for the reference case.	1.0 E-14 cm^2/sec	1.0 E-8 cm^2/sec
	P5: Waste residual – Inventory	1 in. of waste	0.1 in. of waste residual	10 in. of waste residual
Vadose zone	P6: Past leaks depth – 200 East	150 ft bgs	130 ft bgs	170 ft bgs
	P7: Past leaks depth – 200 West	130 ft bgs	110 ft bgs	150 ft bgs
	P8: Past releases – Inventory	Reference case inventory	See discussion in Section 3.5.3.2	See discussion in Section 3.5.3.2
	P9: Unsaturated flow	Variation of unsaturated hydraulic conductivity via K_{sat} (defined for each vadose zone layer).	K_{sat} * 0.1 for each layer	K_{sat} * 10 for each layer
	P10: Uranium K_d	Reference case uranium of 0.6 mL/g	0.2 mL/g	4 mL/g
	P11: Iodine K_d	Reference case iodine of 0.2 mL/g	0.1 mL/g	2 mL/g
	P12: Technetium K_d	Reference case technetium of 0.0 mL/g	0 mL/g	0.1 mL/g
Unconfined aquifer – 200 East WMAs	P13: Hydraulic conductivity	3,000 m/day	2,000 m/day	4,000 m/day
	Effective porosity	0.25	NC	NC
	Hydraulic gradient	0.00001	NC	NC
	Depth to water table	79 m	NC	NC
Unconfined aquifer – 200 West WMAs	P14: Hydraulic conductivity	25 m/day	7.5 m/day	50 m/day
	Effective porosity	0.1	NC	NC
	Hydraulic gradient	0.0005	NC	NC
	Depth to water table	78 m	NC	NC

NC = not considered

* indicates multiply by the number.

3.5.4 Estimated Ranges for Selected Parameters in Groundwater Pathway "What if …?" Cases

The reference case analysis represents the expected performance of the closed tank system. This section examines a number of postulated conditions or alternatives that, though considered less likely than the reference case, could potentially occur. The ability of the system to perform as expected even under these "unlikely" conditions is considered a measure of the robustness of the SST closure system. Should the results of these analyses show an unacceptable degradation of the WMA closure system performance, two options are available:

- The approach to retrieval and/or the design of the engineered system can be changed to provide an additional level of security, or
- Additional characterization of the system may be necessary to better determine the presence or absence of a problematic feature or to better characterize the natural feature leading to the unacceptable performance.

Table 3-15 summarizes the alternative or "what if" conditions examined. The likelihood of each set of scenarios is considered small but nonetheless presents an element under the philosophy of defense in depth that is considered important in investigating the level of protectiveness for the reference case (see defense in depth discussion in Section 1.6).

For the purposes of "what if" simulations, the condition stated in the alternative is addressed keeping all other features of the closure system unchanged. Unless stated otherwise, analyses are conducted on WMAs C and S-SX. The "what if" cases are grouped according to the key barriers and features identified in Section 1.7 as follows:

- Section 3.5.4.1 describes the "what if" conditions for the **surface barrier**
- Section 3.5.4.2 describes conditions for **potential retrieval leaks**
- Section 3.5.4.3 describes conditions for the **closed SSTs**
- Section 3.5.4.4 describes conditions for the **vadose zone**
- Section 3.5.4.5 describes "what if" conditions for the **exposure parameters**.

Table 3-15. Alternatives to the Reference Case or "What if" Conditions for the Examination of the Level of Protectiveness Provided by the Reference Case for the Protection of Groundwater (3 pages)

Barrier/Feature	Alternative	Condition	Description/Action
Surface Barrier	A1	What is impact of closing the farm before year 2032?	An earlier (year 2020) placement of the final closure interim barrier (as opposed to year 2032 for the reference case)/sensitivity case.
	A2	What is the impact of not closing the farms by year 2032?	A later (year 2050) placement of the final closure barrier will be examined/sensitivity case.
	A3	What is the impact of an interim barrier by year 2010 over major leaks?	An interim barrier will be placed over the large leaks in WMAs S-SX and C beginning in the year 2010/sensitivity case.
	A4	What is the impact of episodic infiltration?	The impacts of episodic infiltration are considered sufficiently analyzed in past work by Smoot et al. (1989). The results will be summarized, as appropriate.
	A5	What if the barrier subsides?	Degradation of the effectiveness of the barrier due to localized subsidence. It is believed that any useful analysis of this issue at this time requires a more advanced closure and barrier design conceptualization.
	A6	What if irrigated farming occurs after the loss of passive control (500 years)?	Based on information in Mann et al. (2001), an enhanced infiltration rate of 50 mm/yr is assumed to occur over the closed tank farm with the cover assumed removed. Enhanced infiltration would begin at the end of passive institutional controls/sensitivity case.
	A7	What if the barrier fails at the end of passive controls?	Assume that the barrier fails at the end of passive controls (500 years). Failure is assumed through loss of silt-loam mix and infiltration increases to background of 3.0 mm/yr in the 200 East Area and 4.0 mm/yr in the 200 West Area (Last et al. 2004b)/sensitivity case.
	A8	What if the barrier fails prior to the end of passive controls?	Assume that the barrier fails at the end of 300 years. Failure is assumed through loss of silt-loam mix and infiltration increases to background of 3.0 mm/yr in the 200 East Area and 4.0 mm/yr in the 200 West Area (Last et al. 2004b)/sensitivity case.

Table 3-15. Alternatives to the Reference Case or "What if" Conditions for the Examination of the Level of Protectiveness Provided by the Reference Case for the Protection of Groundwater (3 pages)

Barrier/Feature	Alternative	Condition	Description/Action
Grouted Tank/ Structure	A9a	What if the 100-Series tanks leak during retrieval?	9a: Simulate a retrieval leak loss of 8,000 gal per tank for a 100-Series tank that is assumed to be by the modified sluicing retrieval method/sensitivity case.
	A9b		9b: Simulate a retrieval leak loss of 20,000 gal per tank for a 100-Series tank that is assumed to be retrieved by the modified sluicing retrieval method/sensitivity case.
	A10	What if retrieval leaks occur at the 200-Series tanks, regardless of the use of dry retrieval methods?	Simulate the effects of a 400-gal leak for each 200-Series tank/sensitivity case.
	A11	What if the grout does not provide the level of encapsulation expected?	Conduct a bounding analysis of this situation based on the assumption of an advection-dominated release for residual tank wastes/sensitivity case.
	A12	What if more tank waste residual is left than expected?	This possibility is addressed in the sensitivity analysis of possible ranges of tank residue (Table 3-14).
	A13	What if a retrieval leak occurs over a past leak prior to tank stabilization?	Simulate an 8,000-gal retrieval leak occurring over a past leak.
	A14	What if the tanks behave like a "bathtub" and collect water, which then releases suddenly?	The void space left within the tank after grout fill is minimal that this is considered a highly unlikely scenario and is bounded by other analyses.

Table 3-15. Alternatives to the Reference Case or "What if" Conditions for the Examination of the Level of Protectiveness Provided by the Reference Case for the Protection of Groundwater (3 pages)

Barrier/Feature	Alternative	Condition	Description/Action
Vadose Zone	A15	What if potential preferential paths were missed during characterization?	Incorporate clastic dike effects for the retrieval leak simulation of 8,000 gal for a 100-Series tank that is assumed to be retrieved by the modified sluicing retrieval method.
	A16	What if the groundwater level does not decline as projected?	Simulate the effect by decreasing the vadose zone thickness by 2 m/sensitivity case.
	A17	What if the depths of past leaks were underestimated?	This contingency is addressed in the sensitivity analysis (Table 3-14).
	A18	What if past leak contamination were underestimated?	This contingency is addressed in the sensitivity analysis (Table 3-14).
	A19	What if remediation of up to 50% of past leaks were possible?	Simulate the removal or immobilization of 5%, 25%, and 50% of mobile contaminants from past leaks/sensitivity case.
	A20	What is the effect of assuming anisotropy for the vadose zone geologic units?	Simulate assuming isotropic saturated hydraulic conductivity for the individual geologic units within the vadose zone.

3.5.4.1 Surface Barrier

The first eight alternatives in Table 3-15 examine impacts to the reference case associated with the timing and performance of the surface barrier. Alternatives 1 and 2 address the timing of the placement of a barrier to infiltration. Currently, all WMAs are projected for closure by year 2032 per Milestone M-45-00 as found in the HFFACO (Ecology et al. 1989). The first two alternatives examine the importance of the timing of the final barrier over the entire farm, assuming an early closure in year 2020 and a late closure in year 2050.

The third alternative examines the importance of interim barriers over known past releases. An interim barrier is defined as a temporary barrier that would be installed to protect groundwater resources prior to complete retrieval and final closure of a tank farm. This alternative only postulates such a barrier over limited areas in deference to the potential impacts such a barrier might have on delaying final closure due to interference with field activities. The interim barrier is assumed to be installed over the largest tank leak or spill in WMAs S-SX and C starting in the year 2010.

The impacts of episodic infiltration (alternative 4) were addressed previously in Smoot et al. (1989). Smoot et al. (1989) addressed infiltration of meteoric water through sediments at the SST farms and the impact of this transient infiltration on contaminant plume movement for evaluating alternative remedial actions (i.e., interim barriers) for leaking SSTs at the Hanford Site. The results of this investigation are used to justify the use of temporally averaged infiltration rates for long-term simulations.

Subsidence of the barrier (alternative 5) is not expected to be a significant issue in the closure of tank farms. Large voids will be grouted and the surface cover will be designed to minimize the impacts of belowground subsidence should it unexpectedly occur and alter the performance of the surface barrier. An applicable and useful analysis of subsidence requires the development of a subsidence causing event and the propagation of the effects of such an event to the surface barrier. For such an analysis to have any value in the design process, the closure process including the barrier design needs to progress to a more complete state.

Alternatives 6 through 8 address the impacts to groundwater should the barrier effectively fail to provide the long-term reduced infiltration rates assumed for the closure system. This would require a significant breakdown in institutional controls. Alternatives 7 and 8 assume that such a breakdown occurs in 500 and 300 years, respectively, after closure. Infiltration is assumed to return to background conditions of 4 and 3 mm/yr for the 200 West and East Areas, respectively (GOSPL [Last et al. 2004b]).

Alternative 6 adds the extremely unlikely condition that after the barrier is removed, active farming occurs on the barrier site. Recharge associated with this alternative follows the approach outlined in the 2001 Hanford immobilized low-activity waste PA (Mann et al. 2001) and 200 East and 200 West Area solid waste burial ground PAs (Wood et al. 1995a, 1996). Recharge assumed from irrigated farming is assumed to average 50 mm/yr. This represents an overestimate of the irrigation volume available for infiltration. Historical estimated natural recharge rates range from 0.1 to 10 mm/yr depending on the crop and water management practices. Wine grapes would have the least recharge, while field crops would potentially have the greatest deep percolation losses (recharge). This analysis assumed a 50 mm/yr rate of infiltration that is initiated at the end of passive controls.

3.5.4.2 Potential Retrieval Leaks

Alternatives 9a, 9b, 10, and 13 in Table 3-15 address alternatives to the reference case associated with the retrieval or processing of the tank waste. The reference case assumes that modified hydraulic sluicing will be used to retrieve waste from all 100-Series tanks considered sound. As alternative cases (9a, 9b, and 13), leaks of 8,000 gal, 20,000 gal, and 8,000 gal over a past leak, respectively, are analyzed for each of the 100-Series tanks assumed to be retrieved by modified sluicing methods. Each analysis is conducted separately to allow the impacts of the differing retrieval methods to be analyzed. Tanks in the 100-Series retrieved using the mobile retrieval system are not assumed to leak during retrieval. Alternative 10 considers a 400-gal leak for the 200-Series tanks.

Assumptions and Justifications for Retrieval Leaks. For retrieval leaks, the following simplifying assumptions were made:

- Hypothetical retrieval leaks of 8,000 gal (alternative 9a) and 20,000 gal (alternative 9b) were assumed from each 100-Series SST that is planned to be retrieved using the modified sluicing process. Known or suspected leakers are assumed to be retrieved using a process with limited water usage; therefore, retrieval leaks are not anticipated for these tanks.
- The retrieval leaks are modeled as starting on January 1, 2000, and leaking at a uniform rate for 14 days.
- All the leaked inventory is readily available for transport with the infiltrating water where transport is only limited by chemical adsorption to the soils.

By assuming that a leak occurs from every 100-Series tank planned to be retrieved using the modified sluicing process, it is possible that a maximum waste volume could infiltrate into the vadose zone. The difference between the actual and presently assumed lost waste volumes may be quite large as only a few tanks have given a clear indication of substantial loss of containment. Also, there was no evidence of a leak during the retrieval of C-106 tank waste (Lee 2004).

The simplifications for estimating the contribution to the BTCs for potential retrieval leaks are reasonable for the following reasons:

- The assumption of a lower leak volume (i.e., 8,000 gal) as a representative leak volume is based on estimates assigned to what leak volumes would be detectable during a sluicing retrieval process. Tank liquid level measurement accuracy is estimated to be approximately $\pm$1 in., which corresponds to approximately 3,000 gal. The 8,000-gal estimate was selected to represent detection under conditions where the tank is being sluiced. Such an estimate of 8,000 gal per tank has been used in earlier retrieval PAs (e.g., DOE-RL 1999).
- Assigning a date of January 1, 2000, for the leak is early and results in estimated concentrations from such leaks arriving at the unconfined aquifer earlier than would be estimated with actual leak dates.

- The 14-day leak duration assumed for the simulation is arbitrary; however, given the averaging that occurs in transport through the vadose zone, the resulting impact is relatively insensitive to this parameter.
- As an alternative case to an 8,000-gal leak, leaks of 20,000 gal are analyzed for each of the 100-Series tanks assumed to be retrieved by modified sluicing methods.

The contaminant sources associated with any potential retrieval leaks were assumed to be readily available for transport with the infiltrating moisture; any chemical retardation is modeled by the linear isotherm K_d model discussed in Section 3.2.2.4. The retrieval contaminant inventory is estimated by multiplying the leak volume (Section 3.5.3.2, Tables 3-10 and 3-11) by the estimated contaminant concentration in the transferring tank liquids for each tank. The source term was modeled with the leak starting on January 1, 2000, and the leak occurring at the bottom east corner of tank S-103 for the WMA S-SX calculations. For the WMA C calculations, the leak was assumed to occur at the bottom of tank C-112.

Inventories for Potential Retrieval Leaks. The composition of the fluid assumed to have been lost during the retrieval process was developed from the HTWOS run (Kirkbride et al. 2005). Inventory estimates were developed by multiplying the assumed leak volumes (i.e., 8,000 gal and 20,000 gal) by the projected fluid compositions for each tank retrieved using the modified sluicing technique. Tanks that are known or suspected to leak were assumed to be retrieved using an alternate retrieval process. For these tanks, no retrieval leaks were assumed.

Table 3-16 summarizes the inventory for potential tank retrieval leaks from SSTs in WMAs C and S-SX for selected radionuclides and chemicals. Complete inventories for all contaminants within the residual waste for each tank that was planned to be retrieved using the modified sluicing technique are provided in Appendix C.

Table 3-16. Potential Retrieval Leak Inventories for Hanford Waste Tanks within Waste Management Areas C and S-SX (2 pages)

Tank [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 Ci [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Waste Management Area S-SX												
S-101	2.98E-01	5.11E-05	1.33E+01	8.42E+02	2.91E+02	1.71E+01	2.98E-01	1.59E-03	6.46E+02	4.84E-06	1.05E-06	1.73E-05
S-102	1.02E+00	1.30E-03	1.96E+00	2.49E+03	5.18E+02	1.16E+01	1.02E+00	1.40E-02	1.25E+03	1.41E-02	3.01E-03	8.96E-03
S-103	1.02E+00	1.01E-03	5.19E+00	2.24E+03	5.12E+02	1.01E+01	1.02E+00	8.82E-03	1.21E+03	3.73E-03	7.79E-04	1.56E-03
S-105	1.24E+00	1.22E-03	2.47E+00	5.03E+03	1.06E+02	1.13E+01	1.24E+00	3.87E-03	2.92E+02	4.62E-04	9.37E-05	1.61E-03
S-106	1.61E+00	1.58E-03	2.14E+01	5.72E+03	4.51E+02	1.77E+01	1.61E+00	3.53E-03	1.52E+03	1.73E-03	3.77E-04	7.25E-03
S-107	4.34E-02	2.21E-05	5.51E+00	2.05E+02	1.08E+02	1.71E+01	4.34E-02	3.57E-05	1.83E+02	1.00E-03	2.10E-04	0.00E+00
S-108	1.55E+00	1.52E-03	1.59E+01	4.96E+03	7.04E+02	1.85E+01	1.55E+00	1.07E-03	1.52E+03	7.03E-03	1.50E-03	1.58E-02
S-109	1.46E+00	1.43E-03	6.96E+00	8.75E+03	1.04E+02	1.71E+01	1.46E+00	2.10E-03	2.00E+02	7.54E-03	1.48E-03	2.04E-03
S-110	1.15E+00	1.13E-03	2.89E+01	4.47E+03	2.68E+02	1.74E+01	1.15E+00	3.82E-03	8.82E+02	2.36E-02	4.67E-03	2.23E-02
S-111	1.61E+00	1.60E-03	2.60E+01	4.02E+03	5.71E+02	1.76E+01	1.61E+00	1.04E-02	1.17E+03	6.81E-03	1.42E-03	3.33E-03
S-112	1.00E-02	7.55E-03	4.22E-01	8.39E+01	2.37E+01	1.86E+01	1.00E-02	3.46E-04	4.55E+01	1.37E-02	2.43E-03	1.80E-02
SX-101	1.12E+00	1.35E-03	5.90E+01	5.32E+03	4.45E+02	1.82E+01	1.12E+00	2.22E-03	1.74E+03	2.05E-03	4.18E-04	1.61E-01
SX-102	1.56E+00	1.59E-03	2.36E+01	4.17E+03	1.40E+03	1.94E+01	1.56E+00	9.07E-03	2.72E+03	9.40E-03	1.98E-03	1.12E-02
SX-103	1.36E+00	1.34E-03	7.80E+00	4.78E+03	8.72E+02	1.88E+01	1.36E+00	3.45E-03	1.99E+03	1.88E-02	3.86E-03	1.99E-03
SX-105	1.57E+00	1.55E-03	1.37E+01	3.97E+03	1.32E+03	1.86E+01	1.57E+00	7.79E-03	1.71E+03	9.70E-02	1.89E-02	5.30E-02
SX-106	2.08E+00	2.08E-03	7.26E+00	4.54E+03	1.39E+03	1.93E+01	2.08E+00	7.63E-03	2.34E+03	9.16E-02	1.97E-02	3.26E-02

Table 3-16. Potential Retrieval Leak Inventories for Hanford Waste Tanks within Waste Management Areas C and S-SX (2 pages)

Tank [c]	Contaminants for Groundwater Pathway Impacts [a]					Contaminants for Inadvertent Intruder Impacts [b]						
	Tc-99 Ci [d]	I-129 Ci [d]	Cr kg [d]	NO_3^- kg [d]	NO_2^- kg [d]	Sr-90 Ci [d]	Tc-99 Ci [d]	Sn-126 Ci [d]	Cs-137 Ci [d]	Pu-239 Ci [d]	Pu-240 Ci [d]	Am-241 Ci [d]
Waste Management Area C												
C-102	1.75E-02	2.12E-03	1.11E+00	1.41E+03	4.10E+02	5.43E+01	1.75E-02	1.60E-05	4.20E+02	4.16E-02	9.96E-03	8.31E-01
C-103	5.04E-01	3.26E-03	5.92E-01	4.08E+01	4.18E+02	8.73E+01	5.04E-01	9.35E-05	4.74E+02	4.70E-02	9.90E-03	3.69E-01
C-104	1.42E+00	8.57E-03	3.53E+00	5.26E+02	9.82E+02	1.06E+02	1.42E+00	4.06E-04	1.34E+03	5.76E-02	1.48E-02	3.29E-01
C-107	6.15E-01	1.93E-02	9.21E+00	4.00E+02	9.45E+02	7.67E+01	6.15E-01	5.97E-02	5.87E+02	4.97E-01	8.07E-02	0.00E+00
C-108	2.86E-01	4.98E-05	6.00E+00	8.20E+02	4.55E+02	2.55E+01	2.86E-01	1.11E-05	1.59E+02	3.97E-03	4.31E-04	5.48E-02
C-109	1.37E+00	1.09E-03	5.10E+00	1.01E+03	6.91E+02	3.74E+01	1.37E+00	4.23E-06	8.83E+01	5.26E-02	9.04E-03	1.96E-02
C-112	1.92E+00	8.29E-04	4.00E+00	1.53E+03	1.16E+03	3.66E+01	1.92E+00	1.66E-06	1.98E+03	5.43E-02	6.83E-03	1.97E-01

[a] Major contaminants for groundwater impacts based on estimated chemical distribution coefficients $\leq$ 0.2 mL/g (Section 3.4.4.1.3).

[b] Major contaminants for inadvertent intruder impacts based on major contaminants contributing to the intruder doses in previous analysis (Mann et al. 2001; Mann and Connelly 2003).

[c] Potential tank retrieval leak inventories based on 8,000-gal leak estimate and average tank retrieval liquid concentration from Kirkbride et al. (2005).

[d] Radionuclide inventories in Ci decayed to January 1 2004; chemical inventories in kg.

3.5.4.3 Closed Tank Structure

The reference case assumes a diffusional release model for contaminants from tank residuals. This is a simplified model of tank release that assumes all the residual wastes is available for diffusion, and does not take into consideration the tank itself or chemical interactions of the residue with the waste form. The development of a refined release model and appropriate chemical/waste interaction database is needed to fully address this issue.

Alternative 11 in Table 3-15 addresses the unlikely possibility that the grout fill fails to encapsulate any of the waste and allows water to pass through the waste (i.e., via advective transport). The advection-dominated release model (mixing-cell cascade model) is used to simulate release from unstabilized wastes. For unstabilized wastes, the radionuclides exit the facility at a rate determined by the flow of water and the amount of dispersion (mixing) within the tank. The mixing-cell cascade model (Kozak et al. 1990) is based on the dispersion analysis of chemical reactors, and allows the analysis to incorporate the effects of dispersion within the tank in a simplified manner. In this model, the tank interior is considered to be composed of a cascade of *N* equal-sized, well-stirred cells in series. The total volume of the *N* cells is equal to the volume of the tank residual waste within the mixing zone.

The mixing-cell cascade model for *N* equal-sized cells is described by Equation 3.13:

$$Q(t) = qAC_0 \exp^{-\alpha N t} \sum_{n=1}^{N} \frac{(\alpha N t)^{n-1}}{(n-1)!} \qquad \textbf{Eq. 3.13}$$

where:
Q = release rate (Ci/yr)
t = time (yr)
q = vertical Darcy flux (m/yr)
A = horizontal (planar) area of the tank interior
α = $q/(\theta d R)$
θ = volumetric moisture content in the residual waste
d = vertical mixing depth (m)
R = retardation factor in the waste material (assumed R=1).

For advection-dominated release, backfill (sand and gravel) was used as the tank fill material. The spatially variable velocities, V, and moisture contents, θ, which are obtained via flow modeling within the tank, are used to determine C_0. A vertical mixing length, d, of 0.825 m (same as that for diffusion-dominated release) was assumed.

The initial concentration of contaminant in the interstitial water can be determined from Equation 3.14:

$$C_0 = \frac{m}{\theta V R} \qquad \textbf{Eq. 3.14}$$

where:
m = total facility inventory (assumed unity) of the radionuclides in the tank
V = equals total volume of the residual waste (i.e., 360 ft^3 [10.2 kL] for 100-Series tanks and 30 ft^3 [850 L] for 200-Series tanks or 1% residual following the HFFACO goal [Ecology et al. 1989]).

The mixing-cell cascade model provides results equivalent to that for a one-dimensional, convective-dispersion equation with varying values of the dispersion coefficient (Kozak et al. 1990). In the limit, as N approaches infinity, the model represents flow through a system with zero dispersion, whereas for N equal to one, the model represents flow with an infinite dispersion coefficient. A value of $N = 10$ will be used reflecting moderate dispersion.

Alternative 12 in Table 3-15 addresses the possibility that more tank residual wastes are left behind than the current baseline estimate of 360 ft^3 (approximately 1 in. of waste spread across the bottom of the tank). The impact of this alternative is addressed in the sensitivity analysis examining the range of tank residual wastes (Table 3-14).

Alternative 13 in Table 3-15 addresses the potential impacts should a pressurized water line break over a past release. It is assumed that, as part of good housekeeping, the issue of water line leaks from existing piping and infrastructure at the tank farms will be addressed and resolved. However, as part of the alternatives analysis, a water line leak could occur over a past leak prior to completion of retrieval. The impact of such an event would most likely be seen as earlier impacts on groundwater from the past leak. This analysis is reported in the WMA S-SX FIR (Knepp 2002a).

An alternative conceptualization that has been often proposed is the situation whereby a tank somehow behaves like a bathtub and fills its interstitial volume with water which is then presumed to be released all at once (alternative 14 in Table 3-15). This alternative is expected to release a small pulse of contaminants into the system instead of the more expected condition of release at a slower rate over a longer period. This alternative was posed when there was belief that the tank might be filled with a highly permeable and porous material such as sand. Current plans call for the tanks to be filled with grout, thus eliminating any possibility for a "bathtub" effect to occur. Also, the sensitivity case where the tank is assumed to be filled with backfill material (sand and gravel) and the tank residual waste contaminants are released with the infiltrating moisture provides a relative bounding case with respect to the bathtub effect. This alternative therefore will not be considered further.

3.5.4.4 Vadose Zone

An alternative characterization that potentially impacts the expected performance of the system is the potential unnoticed presence of clastic dikes (alternative 15 in Table 3-15). Clastic dikes are ubiquitous sedimentary structures observed in outcrops and trenches that expose the Hanford formation in the 200 Areas. The dikes are believed to represent dewatering structures that developed during compaction and settling of cataclysmic flood deposits during or soon after floodwaters drained from the Pasco Basin. The dikes are of particular interest because they occur as near-vertical tubular bodies filled with multiple layers of unconsolidated sediments. Simulations (alternative 15) were run to evaluate effects of clastic dikes on the peak concentration at the WMA fenceline.

The reference case assumed that the long-term, post-Hanford unconfined aquifer hydraulic head distribution is representative of the pre-Hanford Site operations condition. Alternative 16 in Table 3 15 (conceptualization) addresses the impacts if the aquifer were to drop by 2 m, thereby increasing the residence time of contaminants within the vadose zone.

Alternatives 17 and 18 in Table 3-15 examine differing conceptualizations regarding the size and location of past releases. This analysis is incorporated into the sensitivity analyses presented earlier. As discussed in Section 3.2, the simulated cases for past releases do not attempt to model a release event itself; instead, they model the potential risk posed by the existing vadose zone contamination footprint. Information on contamination footprints and their location within the vadose zone is based on spectral gamma data for drywells and recently-drilled borehole data in the vicinity of known releases. Modeling results contained in the FIRs (Knepp 2002a, 2002b) have shown that the peak concentrations for contaminant BTCs are influenced more by total inventory (Ci or Kg) than by spatial distribution of that inventory within the vadose zone. These results are also supported by past risk analysis (Jacobs 1998).

Alternative 19 in Table 3-15 is an examination of the impacts of an assumed immobilization of varying levels of mobile contaminants from past releases. No statement concerning the need to remediate is implied.

The individual geologic units within the vadose zone are known to be anisotropic with respect to saturated hydraulic conductivity. Alternative 20 in Table 3-15 is a postulated case that assumes arbitrarily that the vadose zone geologic units are not anisotropic.

3.5.4.5 Exposure Parameters

The reference case considers the estimated impacts to the all-pathway farmer for the groundwater pathway and the driller and post-intrusion rural pasture for the intruder scenario. Estimated impacts for other exposure scenarios are provided in Chapter 6.0 using the dose factors provided in Rittmann (2004).

3.5.5 Intruder Dose – Sensitivity to Parameter Assumptions

Intruder dose comes from inadvertent human intrusion into the waste disposal site after closure (Rittmann 2004). It is assumed that a well is drilled through the waste, bringing some of the waste to the surface where people can be exposed to it.[4] The risk metric for waste intrusion is the EDE from radionuclides in the waste. The projected dose depends first on the assumed exposure scenario because different exposure scenarios lead to different doses. The exposure scenarios are constructed from assumptions about which exposure pathways are applicable and which parameter values are appropriate. There are uncertainties associated with each of the parameters. The objective of this section is to quantify the likely range of each parameter.

The intruder dose depends on two quantities: 1) the soil contaminant concentration that the intruder is exposed to and 2) the exposure scenario dose factor. Parameters used to develop the scenario dose factors will not be discussed here; however, they do depend strongly on the waste composition. In addition, only the reference case intruder scenarios are discussed. The acute scenario is the well driller, while the reference case chronic scenario is the rural pasture with a cow.

For the acute scenario (well driller), the soil contaminant concentration used in the dose calculation is the average in the borehole cuttings. For the chronic scenarios (rural pasture),

[4] Intruder dose metrics are derived from 10 CFR 61 regulations and do not correspond with all-pathways exposure scenarios associated with groundwater.

the soil contaminant concentration used in the dose calculation is the average in the contaminated pasture. The soil contaminant concentration depends on the factors listed in Table 3-17. The waste composition is not listed because the relative amounts of various radionuclides can vary widely.

Table 3-17. Intruder Pathway – Summary of Reference Case Parameters and Expected Ranges

Parameter	Parameter Range		
	Minimum	Reference	Maximum
1. Waste concentration at closure	0.1 [a]	varies	10 [a]
2. Decay time at intrusion	100 yr	500 yr	1,000 yr
3. Waste thickness – tank residual	0.004 m	0.025 m	0.15 m
4. Waste thickness – unplanned releases	0.25 m*	varies	4 m*
5. Fraction available for internal dose (residual tank waste only)	0.01	0.1	1
6. Borehole depth – acute	70 m	80 m	90 m
7. Borehole diameter – chronic	8 in.	10 in.	12 in.
8. Spreading area – chronic	3,000 m^2	5,000 m^2	7,000 m^2
9. Tilling depth – chronic	0.1 m	0.15 m	0.20 m

[a] Fraction of reference case value representing minimum or maximum value for range.
* indicates multiply by the number.

Table 3-17, row 1, labeled “Waste concentration at closure” is the activity per unit volume or mass of the various radionuclides in a tank or UPR. The present inventory is the best estimate, but it may be larger or smaller by an order of magnitude. The inventory estimates err on the high side, so the largest concentration is unlikely to exceed a factor of 10 greater than the estimates used.

Row 2 labeled “Decay time at intrusion” is the time between site closure and intrusion. The range shown comes from DOE M 435-1.1 as the time period of interest for inadvertent intrusion.

Rows 3 and 4 address the assumed waste thickness. Two rows are needed because residual waste and the UPRs to soil have different uncertainties. The residual waste in the tanks has an average thickness of 1 in. or less. Due to the shape of the bottom of the tank and the difficulty in removing some attached solids, the waste thickness will vary. Based on the shape of the tank bottom, the amount of waste exhumed could be larger by a factor of 6 if the intruder’s well passes through the center of the tank. The intruder dose varies linearly with the thickness of the residual tank waste.

The thickness of UPRs to the soil depends on the horizontal spread of the plume. As discussed earlier, releases can lead to significant horizontal spreading (Section 3.2.2). The reference case assumes that the plume has a diameter equal to its height. The relative height and width of the plume could vary by a factor of 5, based on observed plumes near the underground tanks that have leaked in the past. Another consideration is the volume of the soil compared to the volume of liquid that entered the soil. The reference case assumes that the contaminated soil has a volume 10 times the volume of the liquid. The likely range for the soil-filling fraction is 5% to 15% based on soil porosity and residual moisture content. The combination of these ranges leads to a waste thickness that may vary by a factor of 4 from the reference case.

The fraction of the exhumed waste that is available to produce internal dose (row 5) is important because the external pathway may sometimes be a small contributor to the intruder dose. For these cases, the intruder dose is roughly proportional to the fraction available for internal dose. When the external pathway is important, there is a sublinear relationship between this fraction and the intruder dose. The fraction available is estimated to range from 1% up to a maximum of 100%. Since the fraction used in the intruder calculations is 10%, the fraction available could be larger or smaller by a factor of 10. The intruder doses will vary by nearly the same factor.

The borehole depth (row 6) is important for the well driller scenario (acute). The ratio of waste thickness to borehole depth determines the waste dilution. The uncertainty in this parameter is small. The depth to groundwater is known and unlikely to change significantly in the future.

The borehole diameter (row 7) is important for the rural pasture scenario (chronic). The volume of waste exhumed depends on the cross-sectional areas for the well. The uncertainty in this parameter is small. The well diameter is based on current drilling practices near the Hanford Site.

Rows 8 and 9 indicate the volume of soil into which the exhumed waste is diluted. The dilution volume is the product of the spreading area and the tilling depth. Note that the pasture area is much larger than the likely spreading area for the borehole cuttings. The cow forages over the well cuttings and elsewhere in the pasture until it obtains the amount of food it eats in a year. The milk concentration varies during the year, but the average is proportional to the average soil concentration in the pasture.

Note also that if the spreading area (row 8) changes appreciably, other exposure parameters must change also. Smaller spreading areas lead to reduced contact with the contaminants. They require less attention, so the individual spends less time in the contaminated area and therefore receives smaller external doses. The individual also inhales and ingests less contaminated dust. For the pasture scenario, the spreading area is driven by the caloric intake for the cow. Reducing the area of the pasture means higher concentrations in the pasture grass, but the cow eats less from the pasture and obtains food elsewhere. Hence, the spreading area is assumed to vary by no more than the range shown in Table 3-17.

The tilling depth (row 9) is also related to the thickness of soil from which grasses derive nutrients. If the tilling depth is much smaller, the soil concentration is larger, but the plants obtain a portion of their nutrients from uncontaminated depths of the soil. Hence, the tilling depth is assumed to vary by no more than the range shown in Table 3-17.

3.5.6 Air Pathway Risk – Sensitivity to Parameter Assumptions

The air pathway addresses volatile contaminants remaining in the closed disposal system and their migration through the grouted tank structure and surface cover. An examination of this pathway leads to the following observations supporting the use of a bounding analysis:

- Few contaminants in the waste are volatile.
- For the important volatile contaminants (tritium, carbon-14, and radon-222), estimations of remaining inventory indicate small quantities will remain in the tank waste at closure.
- Very low human exposure impacts are estimated under credible exposure scenarios.

The low human exposure impact considered possible through this pathway, even under an extreme set of bounding conditions (Section 3.2.3), does not warrant a more complicated analysis examining features and processes of the release mechanism for vapors.

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