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Pacific Northwest National Laboratory
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Recharge Data Package for Hanford Single-Shell Tank Waste Management Areas

M. J. Fayer
J. M. Keller

September 2007

Prepared for CH2M HILL Hanford Group, Inc.
and the U.S. Department of Energy
under Contract DE-AC05-76RL01830

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Richland, Washington 99352

Preface

The borehole data considered for this data package are current as of March 31, 2007. The meteorological data summary information in Section 3 is current through December 2004. The daily meteorological information used in Appendix C is current as of December 31, 2006. Unless noted, the field measurement data are current through March 2004.

All averaging (e.g., chloride concentrations; multiple recharge estimates) is arithmetic unless noted otherwise.

The precision and accuracy of the recharge estimation methods vary. Rather than reconcile the differences, this report presents all recharge estimates with two significant digits, recognizing that each method has technique-specific uncertainties.

Summary

Pacific Northwest National Laboratory (PNNL) is assisting CH2M HILL Hanford Group, Inc., in its preparation of the Resource Conservation and Recovery Act (RCRA) Facility Investigation report. One of the PNNL tasks is to use existing information to update and supplement the existing estimates of recharge rates for past and current conditions as well as future scenarios involving cleanup and closure of tank farms. The existing information includes recharge-relevant data collected during activities associated with a host of projects, including those of RCRA, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), the CH2M HILL Tank Farm Vadose Zone Project, and the PNNL Remediation and Closure Science Project. As new information is published, the report contents can be updated.

The objective of this data package was to use published data to provide recharge estimates for the scenarios being considered in the RCRA Facility Investigation for the single-shell tank (SST) waste management areas (WMAs). Recharge rates were estimated for areas that remain natural and undisturbed, areas where the vegetation has been disturbed, areas where both the vegetation and the soil have been disturbed, and areas that are engineered (e.g., surface barrier). The recharge estimates update and supplement the estimates provided by Last et al. (2006) for the Hanford Site using additional field measurements and simulations using weather data through 2006.

Table S.1 shows the recommended recharge rates for various soil and vegetation combinations. Values in the table that differ from those provided by Last et al. (2006) are highlighted. The largest changes are to the rates for shrub communities. Those changes were brought about by the addition of more borehole tracer data, including the boreholes at the Integrated Disposal Facility (IDF).

Table S.1. Estimated Long-Term Drainage Rates for Use in Hanford Assessments. Items in bold indicate supplements or revisions to recommendations by Last et al. (2006).

	Estimated Long-Term Drainage Rates (mm/yr)	
Soil Type	Shrub	No Plants
Rupert sand (near U.S. Ecology)	5.0	30
Rupert sand (near IDF)	0.9	**45**
Rupert sand (elsewhere on Central Plateau)	**1.7**	**45**
Burbank loamy sand	**1.9**	**53**
Ephrata sandy loam	**2.8**	**23**
Hezel sand	**<0.1**	**8.7**
Esquatzel silt loam	**<0.1**	**8.6**
Hanford formation sand	np	**62**
Graveled surface	np	92
Modified RCRA C barrier	0.1	0.1
Gravel side slope on surface barrier	**1.9**	33[a]

np = Not provided by Last et al. (2006) or this data package.
(a) Tentative because of concerns regarding presence of some plants, road, and small section of vegetated silt-loam within the drainage collection zone.

One key conclusion important for evaluation of recharge in SST WMAs is that the data in the borehole characterization reports to date are not sufficient for recharge estimation. Recharge evaluation requires specific data types, spatial resolution, measurement techniques, and system knowledge. Because recharge estimation was not identified as a critical need during the data quality objective (DQO) process, these requirements for recharge estimation were not considered when the SST borehole characterization efforts were planned and executed.

A key issue identified in this study was the lack of any physical and hydraulic characterization data for the near-surface material in SST WMAs. These data are needed to predict recharge rates for each SST WMA because it is visually clear that the surface material differs from WMA to WMA. The data are needed also to be able assign recharge rates based on lysimeter data for specific materials. Sampling within WMAs is recognizably problematic, but it can be done (just as borehole samples were collected from deeper depths for geochemistry studies).

Given the current state of the tank farms and the likelihood that they will look nearly the same for the next 30 years, the rates for unvegetated Hanford sand will continue to be important. However, Hanford sand is too non-specific a term to describe the surface sediments at all WMAs. A better method of differentiating the material on a site-specific basis is warranted. Furthermore, the impact of gravel additions as part of the controlled, clean, and stable program must be considered.

Finally, little is known about the speed and character of restoration, particularly with respect to the soils that have been extremely disturbed. We need to understand and be able to predict with more accuracy how disturbed and graveled soils alter recharge rates, how easily they allow vegetation to reestablish, and how soon they begin to develop soil properties such that they start to resemble natural soils. We also need to determine the potential for other processes (e.g., sand dune migration into WMAs) to occur that could affect conditions in the WMAs. If such processes are sufficiently likely, we need to be able to predict how the changes caused by those processes will impact recharge rates.

This report was prepared with full knowledge that its recommendations would evolve as new data and understanding emerged as the facility investigation reports are addressed and closure plans developed. The format of the report is in line with the Hanford Site data package such that the two are complementary.

Acknowledgments

The authors wish to acknowledge F. M. Mann at CH2M HILL Hanford Group, Inc. (Richland, Washington) for providing project funding and technical guidance. The authors greatly appreciate the helpful comments provided by the reviewers, Dr. Bridget Scanlon of the Bureau of Economic Geology at the University of Texas at Austin and Dr. Glendon Gee at Pacific Northwest National Laboratory. We are also appreciative of the review provided by Ken Krupka, the PNNL project manager. We are particularly grateful to Andrea Currie (PNNL) for completing the editorial review and Lila Andor and Mike Parker (PNNL) for formatting of this technical report. Finally, the authors acknowledge the many DOE, Laboratory, and contractor staff whose pursuit of understanding of the environment provided us the data and knowledge that made this data package possible.

Abbreviations and Acronyms

BWTF	Buried Waste Test Facility
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act*
CH2M HILL	CH2M HILL Hanford Group, Inc.
CMB	chloride mass balance
CRBG	Columbia River Basalt Group
DOE	U.S. Department of Energy
DQO	data quality objectives
FIR	field investigation report
FLTF	Field Lysimeter Test Facility
HMS	Hanford Meteorological Station
IDF	Integrated Disposal Facility
ILAW	immobilized low-activity waste
LERF	Liquid Effluent Retention Facility
PNL	former abbreviation for Pacific Northwest National Laboratory
PNNL	Pacific Northwest National Laboratory
RCRA	*Resource Conservation and Recovery Act of 1976*
RFI	RCRA Facility Investigation
SST	single-shell tank
SWL	Solid Waste Landfill
VZTF	Vadose Zone Test Facility
WMA	waste management area

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1.0 Introduction

The Hanford Federal Facility Agreement and Consent Order (Ecology et al. 1989) Milestone M-045-55 requires that a Resource Conservation and Recovery Act (RCRA) Facility Investigation report be submitted to the Washington State Department of Ecology. The RCRA Facility Investigation report, which is being prepared by CH2M Hill Hanford Group, Inc. (CH2M HILL), will provide a detailed description of the state of knowledge needed for tank farm risk assessments. Achieving this goal will require predictions of contaminant migration in and around each tank farm. To make such predictions will require estimates of the fluxes of water moving through the sediment within the vadose zone around and beneath each tank farm. Those fluxes, loosely called recharge rates, are the primary mechanism for transporting contaminants to the groundwater. This document provides the detailed technical information about recharge to support the RCRA Facility Investigation report.

Pacific Northwest National Laboratory (PNNL) assists CH2M HILL in its preparation of the RCRA Facility Investigation report. One of the PNNL tasks is to use existing information to estimate recharge rates for past and current conditions as well as future scenarios involving cleanup and closure of tank farms. The existing information includes recharge-relevant data collected during activities associated with a host of projects, including those of RCRA, the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), the CH2M HILL Tank Farm Vadose Zone Project, and the PNNL Remediation and Closure Science Project. As new information is published, the report contents can be updated.

Although the area of interest is the Hanford Site Central Plateau, most of the information discussed in this document is centered on providing recharge estimates for individual single-shell tank (SST) Waste Management Areas (WMAs). Figure 1.1 shows the location of the Hanford Site within the state of Washington and relative to local features such as the Columbia River and the Rattlesnake and Saddle mountains. The focus of this report is the Hanford Central Plateau and, more specifically, the two main processing areas on the plateau, i.e., the 200 East Area and the 200 West Area. Figure 1.2 shows a more detailed view of the 200 East and West Areas to highlight the locations of the seven SST WMAs.

1.1 Objective

The objective of this data package is to provide recharge estimates for the scenarios being considered in the RCRA Facility Investigation. Recharge estimates are needed for areas that remain natural and undisturbed, areas where the vegetation has been disturbed, areas where both the vegetation and the soil have been disturbed, and areas that are engineered (e.g., surface barrier). Three recharge estimation techniques were used to satisfy the objective: lysimetry, tracer studies, and computer simulation studies. The recharge estimates supplement the estimates provided by Last et al. (2006) for the Hanford Site using additional field measurements, tracer data collected in tank farms, and simulation analysis using weather data through 2006. The data package identifies how the data were used to generate recharge rate estimates. This data package uses the structure of the Integrated Disposal Facility (IDF) recharge data package (Fayer and Szecsody 2004) and retains some of the written material from that data package.

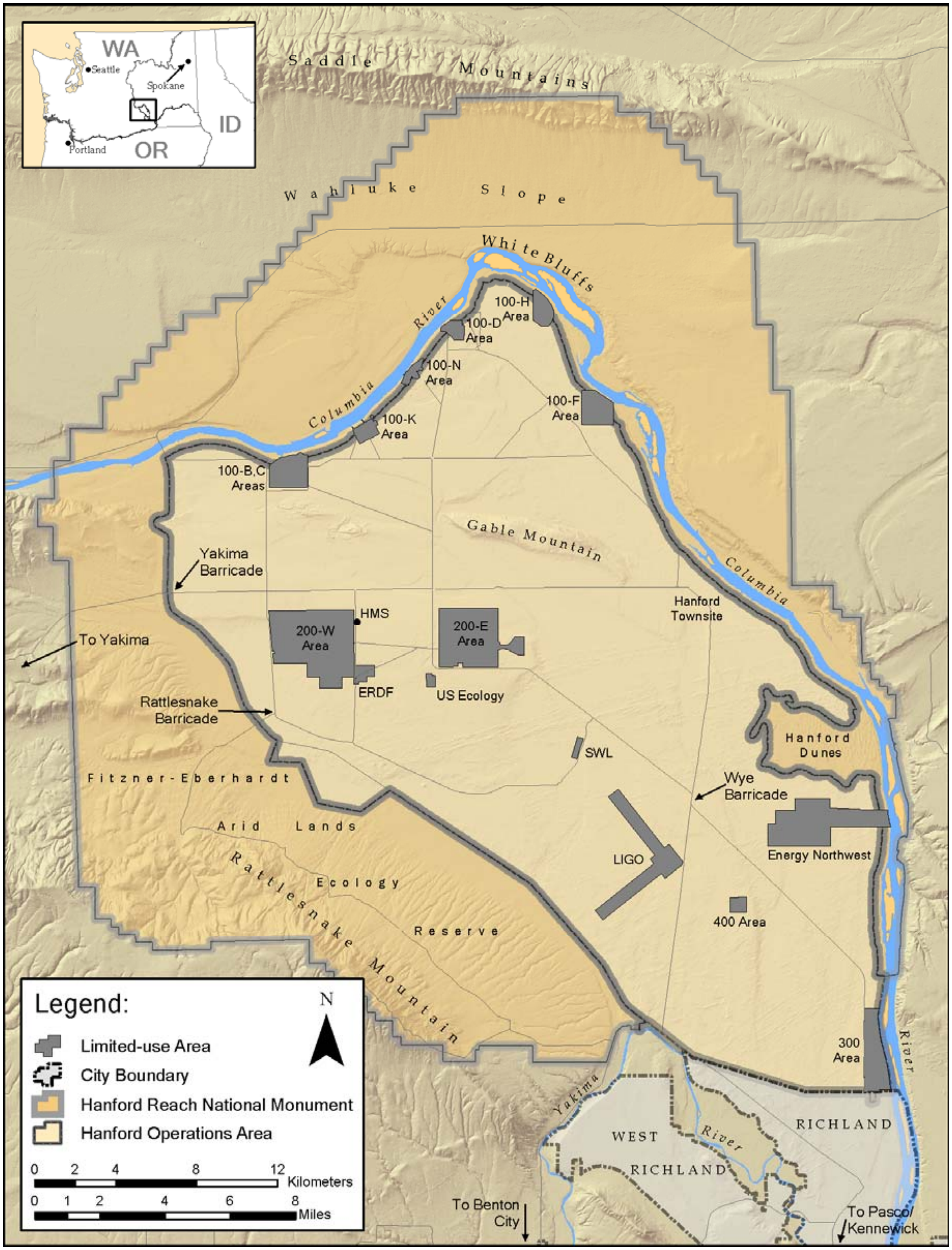

Figure 1.1. Hanford Site Location Within Washington State

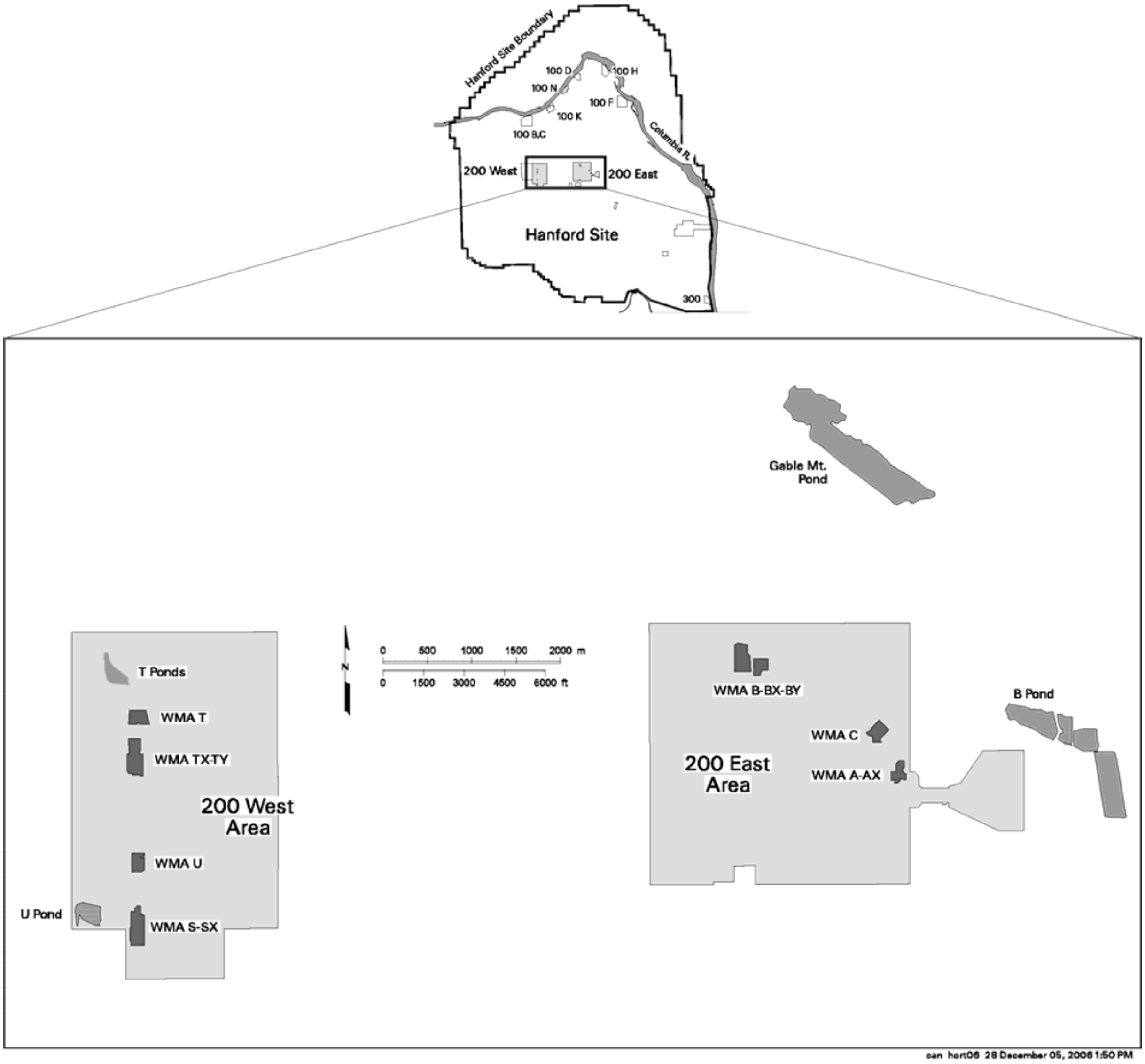

Figure 1.2. Single-Shell Tank Waste Management Areas in the 200 Areas on the Hanford Site (after Horton 2007)

1.2 Challenges

Creating a recharge data package for the entire Central Plateau faces some key challenges that include the extreme sparseness of available data, the lack of direct measurements that can be corroborated (e.g., from multiple methods), and uncertainty regarding past, present, and future climate, vegetation, and land use.

The first challenge involves the lack of data, which is understandable, given the time and expense required to collect it. A single borehole on the Central Plateau can cost several hundred thousand dollars just to drill; environmental tracer analysis costs are extra. That single borehole yields a single estimate of recharge. Drilling additional boreholes in the same general area to estimate recharge variability then becomes recognizably cost-prohibitive for most needs.

The second challenge involves the lack of confirmatory measurements. Previous efforts, as well as this one, use multiple methods to estimate recharge. Unfortunately, the methods measure or estimate recharge for time frames that do not overlap. For example, environmental tracers reflect historical conditions, whereas direct measurements reflect current conditions. The result is that, for many existing recharge estimates, no confirmatory data or estimates are available.

The third challenge is the uncertainty of past, present, and future conditions that affect recharge rates. While the reader probably recognizes and acknowledges the uncertainty of past and future conditions, what may be less well recognized and appreciated is that present conditions (e.g., 1944 to 2007) can encompass just as much uncertainty. For example, soil conditions were extensively reworked during tank farm construction such that existing soil conditions must be inferred. More important, construction and operation activities occurred that have implications for recharge estimation. These include additions of water (e.g., compaction; dust control; runoff from roads and parking lots; water line leaks), emissions of chemicals (e.g., coal-fired power plants; water treatment facilities), and additions of chemicals (e.g., sprays for weed control; spills during transfers or from leaks). For the most part, these activities and events are undocumented.

Each of the above challenges is recognized. While each cannot be overcome completely, progress can occur by relying on well-reasoned estimates based on available data and collecting new data for conditions for which recharge estimates are deemed critical.

1.3 Scope

The scope of this report is limited to providing estimates of recharge rates for SST WMAs on the Central Plateau for conditions prior to Hanford, during Hanford operations, during the surface barrier phase, and subsequent to the design life of surface barriers. The authors acknowledge that a number of other processing and disposal facilities reside on the plateau and are not considered explicitly in this report. Fortunately, many of the recharge estimates that are provided are relevant to those other facilities. More important, this report can be expanded easily to include those other facilities as they are evaluated and deemed important enough to warrant investigation of site-specific recharge rates.

The focus of this report is on natural recharge rates that result directly from precipitation. The recharge rate estimates provide here do not account for localized recharge events such tank leaks or disposal of liquids to ground (e.g., trenches, cribs, reverse wells). They also do not account for runoff events caused by facility features such as roads, parking lots, and compacted soil areas, or for water line leaks or building roof drip lines. The report does not address the variability in recharge estimates, evaluate sensitivity tests, or consider alternative conceptual models. All are valid and appropriate activities that can and should be addressed in subsequent efforts.

Data-gathering for this report was limited to existing publications and datasets as much as possible. By the time this report is published, additional data are likely to be available and certainly more will follow. The recommendations in the report can be reviewed and updated as needed and appropriate.

Following the Introduction in Section 1, the main body of the report contains four sections. Section 2 provides background material concerning recharge. Section 3 presents an overview of the environmental setting to provide context for the recharge estimation and recommendations to follow. Section 4 describes the SST WMAs with respect to conditions that affect recharge. Section 5 describes the recharge estimation methods. Section 6 explains how recharge rate estimates were assigned. Section 7 summarizes the data package.

Three appendices are included. Appendix A summarizes the relevant field measurements used to estimate recharge. Appendix B summarizes the published tracer-based estimates of recharge and evaluates borehole data from tank farms to elicit additional tracer-based recharge estimates. Appendix C summarizes the recharge simulations that were conducted to support the recommendations in this report.

2.0 Recharge Background

The Hanford Site was established in 1943 as a U.S. government nuclear materials production facility. During its history, the site mission included nuclear reactor operation, storage and reprocessing of spent nuclear fuel, and management of radioactive and hazardous waste. The years of operations resulted in the accumulation of significant quantities of radioactive and hazardous waste as well as their intentional and unintentional release to the environment. Today, activities on the Hanford Site involve environmental restoration, energy-related research, and technology development. One of the restoration activities is to plan and conduct remediation and closure of the SST WMAs. This activity will need estimates of recharge rates. This section defines recharge as it is used for this data package, illustrates why the recharge rates are so important, and briefly summarizes the recharge estimates available for use in the RCRA Facility Investigation report.

2.1 Definition

The precise definition of recharge is that flux of water reaching (i.e., recharging) the water table. There is no effective way to measure recharge at the water table beneath the Central Plateau, given the inaccessibility (depth ≥ 80 m); influence of operations (e.g., discharges, remediation pumping); and multiple contaminant plumes. Instead, measurements in the shallow portion of the vadose zone and numerical and tracer analyses are used to estimate the deep drainage flux, i.e., that flux leaving the evapotranspiration zone and ostensibly traveling to the water table. Quantifying this flux is crucial to Hanford Site cleanup because it is this flux of water that can mobilize and transport contaminants in the vadose zone.

Given sufficient time, the deep drainage flux will eventually manifest itself as the recharge flux. However, when deep drainage fluxes change, the change may not be manifested at the water table for hundreds to thousands of years. The length of time for the change to propagate through the vadose zone will depend on the thickness and hydraulic properties of the vadose zone and the initial and final deep drainage rates. Sediment stratification can lengthen that time further.

One point of confusion regarding recharge concerns the oft-repeated comment that subsurface tanks enhance recharge because they divert water along their sloping upper surfaces (e.g., CH2M HILL Hanford 2002). Buried items such as tanks can influence recharge rates if they cause water to remain within the evapotranspiration zone longer than it would otherwise. Under such a scenario, the evapotranspiration rate would be higher and thus recharge would be lower. However, once water has moved below the evapotranspiration zone, an SST may redirect the flow of water laterally but has no effect on the total recharge beneath the WMA. In other words, the average recharge beneath a WMA remains constant, but the distribution of flux within that WMA can become variable.

Plants are prevented from establishing on tank farm surfaces, which means that the evapotranspiration zone is determined solely by the depth to which evaporation operates. The shallowest point on all SSTs with respect to the ground surface is the center, which can be 1.8 to 2.7 m deep. Away from the center, the tank surfaces slope downward to the edge such that the depth of soil above the edge is about 3.6 to 5.4 m. Given the coarse texture of the surficial sediments, we expect the depth of significant influence of evaporation to range from 1 to 2 m, depending on soil hydraulic properties (cf. Appendix C). This suggests that the physical presence of the tanks should not significantly decrease recharge, but given the potential for spatial variability in soil properties, this assumption needs to be confirmed.

2.2 Importance

As noted in Section 1, the deep drainage flux (i.e., recharge) is the primary mechanism for transporting contaminants to the groundwater. Simulation results that supported each of the completed SST field investigation reports (FIRs) (WMA S-SX, CH2M HILL Hanford 2002; WMA B-BX-BY, Knepp 2002; WMAs TX and T-TY, Myers 2005) confirmed that predictions of future contaminant behavior and groundwater impact were most sensitive to recharge and contaminant inventory. For example, Myers (2005) showed that as recharge rates were reduced, conservative solute travel times increased and peak concentrations were reduced anywhere from 88 to 97%. Of course, these results are specific to the conditions evaluated by Myers, but they are consistent with observations in the other FIRs. For each FIR, interim barriers were evaluated as a means to reduce recharge prior to closure; the value of such covers was tank-farm–dependent.

In addition to the direct impact to transport rates, recharge also can affect contaminant mobilization. For example, Bacon and McGrail (2002) demonstrated the importance of recharge by showing how it affected the performance of buried immobilized low-activity waste (ILAW) glass. They evaluated the release of technetium-99 from the ILAW glass when subjected to five different recharge rates. Figure 2.1 shows that the predicted technetium-99 flux in the vadose zone 15 m beneath the ILAW disposal zone was most sensitive to the recharge rate when rates are less than 10 mm/yr. For example, lowering the recharge rate from 4.2 mm/yr to 0.9 mm/yr reduced the predicted technetium-99 flux from 0.6 to 0.008 ppm/yr, a sixteenfold reduction. Such high sensitivity demonstrates the importance of estimating the recharge rate as accurately as possible.

2.3 Previous Estimates

In the early years of the Hanford Site, the perception was that recharge occurred only along the upper elevations of Rattlesnake Mountain and the valleys to the north, and it did not occur across the remainder of the Site. The Hanford Defense Waste Environmental Impact Statement assumed that natural recharge was essentially zero in and around the storage and disposal areas (DOE 1987). A panel of nationally recognized scientists was convened in 1985 to discuss the recharge issue (Gee 1987). The reviewers disputed the notion of zero recharge. Data collected before and after the 1985 review showed clearly that recharge can and does occur under certain soil and plant conditions. Gee et al. (1992) presented evidence that recharge rates can vary from nearly zero in silt loam soil covered by sagebrush to more than 100 mm/yr in gravel-covered soil without vegetation.

Smoot et al. (1989) used numerical simulations to examine the impact of recharge on the contaminant plume that originated from a leak in T farm in 1973. Using a combination of measured and estimated weather variables and soil parameters, they simulated a recharge rate of 131 mm/yr (77% of the precipitation). Smoot et al. (1989) used that rate in a 2-dimensional computer code and simulated the fate of the contaminant plume. Smoot et al. (1989) also conducted additional simulations to demonstrate that alteration of the surface soil properties could significantly reduce the recharge rate (and thus reduce contaminant movement).

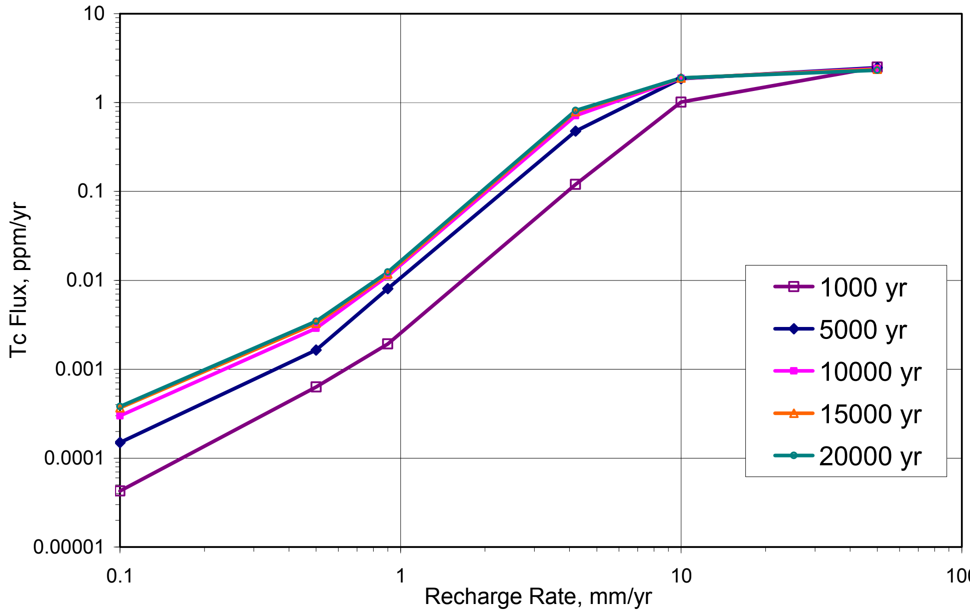

Figure 2.1. Predicted Technetium-99 Flux in the Vadose Zone 15 m Beneath the Immobilized Low-Activity Waste Disposal Zone at Selected Times as a Function of Recharge Rate (adapted from Bacon and McGrail 2002, Figure 5)

Bauer and Vaccaro (1990) prepared a recharge map for the Hanford Site using what they called the deep percolation model and simulation cells with an area roughly 1 km^2. The estimated recharge rates ranged from 0 to 13 mm/yr across the Central Plateau. Their study did not address the impacts of Hanford operations on soil and vegetation conditions, which explains their low estimates for recharge, particularly in the tank farms.

Fayer and Walters (1995) and Fayer et al. (1996) prepared a recharge map for the Hanford Site using a combination of lysimeter measurements, tracer estimates, and the recharge code UNSAT-H. The recharge estimates were assigned to cells with an area of 0.0025 km^2 (50 by 50 m). The authors accounted for disturbance by assuming that no vegetation was present; however, they did not address soil changes (e.g., gravel covers on SST farms). Recharge rate estimates ranged from 2.6 to 55 mm/yr, depending on soil and vegetation conditions.

Rockhold et al. (1995) reviewed previous work related to recharge for a performance assessment project involving an early version of the IDF. Appendix B of their report describes the numerous studies conducted since 1969 using field measurements of soil water, matric potential, and temperature; tracer measurements; lysimeter measurements; and numerical simulation. All of these studies showed the potential for recharge exists and that the rate depends on soil conditions and plant communities. For example, other things being equal, rates are higher for coarse-textured rather than fine-textured soil, for sparse plant communities rather than dense communities, and for shallow-rooted rather than deep-rooted plants. Rockhold et al. (1995) presented the following recommendation:

> We recommend assuming a recharge rate of 0.5 mm/yr through the Hanford protective barrier. This assumption is supported by an 8-year record of lysimeter data (Table 3.1) and is consistent with engineering design specifications over the 1,000-year design life of the barrier (Wing 1994).

At the barrier edge, a higher recharge rate of 75 mm/yr should be assumed. This assumption is based on four years of data for a lysimeter with a graveled surface (Table 3.1) that is similar to the riprap side slope of the protective barrier. This estimate does not include possible overland flow or lateral drainage from the barrier. Beyond the barrier, the recharge rate of the natural ecosystem can be represented with one of two rates. If the plant community is assumed to be sagebrush, an estimate of 5.0 mm/yr should be used. This is a conservative value chosen to be slightly greater than all the rates reported by Prych (1995) using tracer measurements. If the plant community is assumed to be cheatgrass, an estimate of 25.4 mm/yr should be used. This value is based on an 8-year record of water content observations at the Grass Site in the 300 Area (Fayer and Walters 1995). For the entire Hanford Site, we recommend using the recharge distribution map reported by Fayer and Walters (1995).

Rockhold et al. (1995) recommended a set of measurement and simulation activities to improve recharge estimates for the IDF site. A panel of nationally recognized scientists was convened in 1995 to review the needs of the IDF Project (formerly the ILAW Project) for recharge information. The panel concluded that site-specific data would be needed to provide technically defensible estimates.[a] They supported efforts to use lysimetry, tracers, and modeling. The panel noted that the results might not change the recharge estimates significantly but would strengthen the technical credibility of the final recharge estimates used in the performance assessment. The panel also cautioned that uncertainty in conceptual models and supporting data should not be ignored.

Subsequent to the reports by Rockhold et al. (1995) and Fayer and Walters (1995), two recharge data packages were issued with data specifically from the IDF site in the southeastern portion of the 200 East Area. The first was Fayer et al. (1999), which provided recharge estimates for IDF-pertinent scenarios for conditions prior to Hanford, during operations, during the barrier design life, and after the barrier design life. They also evaluated the sensitivity of the recharge estimates to soil and plant properties. In addition, they examined the impact of alternative conceptual models, including altered climate, complete replacement of the shrub community with cheatgrass, and farming directly on the barrier. Key recommended recharge rates were 0.1 mm/yr for the surface barrier with silt loam soil (both during and after its design life), 0.9 mm/yr for undisturbed Rupert sand with a shrub-steppe plant community, 4.2 mm/yr for undisturbed Burbank loamy sand with a shrub-steppe plant community, and 55.4 mm/yr for unvegetated Hanford formation sand such as occurs when the surface soil is stripped off prior to excavation.

The second IDF data package was Fayer and Szecsody (2004), and it essentially updated the previous data package with two additional recharge estimates from IDF boreholes, a longer record of measurements from the lysimeters, and additional simulation sensitivity tests. Recharge recommendations for the barrier, Rupert sand, and Hanford formation sands remained unchanged. The recommended rate for Burbank loamy sand at the IDF site was reduced from 4.2 to 0.9 mm/yr based on IDF-specific data and a recognition that soil type at the IDF site is unique and not readily classified as either Rupert sand or Burbank loamy sand. This recharge rate does not apply to Burbank loamy sand elsewhere unless site-specific soil conditions are confirmed as being similar to those at the IDF site. The fact that soil conditions can be significantly different than predicted by the Hajek (1966) soil map suggests that SST WMA evaluations ought to confirm site-specific soil conditions to support recharge estimation. Fayer

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(a) Honeyman JO. 1995. Letter to L Erickson transmitting the results of the 1995 workshop titled *Summary of peer review comments resulting from the second Hanford groundwater recharge workshop*. May 22–23, 1995, Richland, Washington.

and Szecsody (2004) also revised their estimate of recharge beneath the armored (i.e., graveled) side slope of a surface barrier. The recommended recharge rate was reduced from 50 mm/yr always (Fayer et al. 1999) to 22.3 mm/yr for the first 16 years and 4.2 mm/yr thereafter, assuming shrub-steppe community had developed by that time.

Last et al. (2006) provided a suite of recharge (deep drainage) estimates for use in Hanford assessments. The recharge estimates were derived from a suite of available field data and computer simulation results. The estimates do not account for overland flow from roadways or roofs, subsurface drainage from water line leaks, or any other manmade additions of water, the impacts wrought by future climate change or land-use alterations, variations within soil types, or dune-sand deposition. The estimates were developed for fairly large geographic areas and may not represent the local recharge rates at specific locations. Estimates were provided for natural and disturbed soils and for surface barriers for each of four periods: pre-operations, operations, post-remediation, and final state. The conditions during these periods include natural soil and shrub-steppe plant communities, disturbances that alter the surface soil and vegetation, emplacement of surface barriers, and long-term changes that occur as the waste sites stabilize and return to natural conditions. Last et al. (2006) described the probability distributions of the recharge estimates used to represent the expected range of recharge rates. They provided a method to examine the impact of surface barrier side slopes and the terrain surrounding surface barriers, both of which could significantly affect waste release and vadose zone transport.

Last et al. (2006) recommended recharge rates for all soil types, surface conditions, and several vegetation conditions for all waste sites at Hanford. Table 2.1 shows the key recommendations for the main soil types that will be discussed later in this report. In lieu of having sufficient data to determine probability distributions, Last et al. (2006) recommended using a simple approach in which the standard deviation was approximated as half the mean value (e.g., the values in Table 2.1) and the range was bounded by one-half and double the estimated rate.

Table 2.1. Estimated Long-Term Drainage Rates for Use in Hanford Assessments (after Last et al. 2006)

Soil Type	Estimated Long-Term Drainage Rates (mm/yr)	
	Shrub	No Plants
Rupert sand (near U.S. Ecology)	5.0	30
Rupert sand (near IDF)	0.9	44
Rupert sand (elsewhere)	4.0	44
Burbank loamy sand	3.0	52
Ephrata sandy loam	1.5	17
Hanford formation sand	np	63
Graveled surface	np	92
Modified RCRA C barrier	0.1	0.1[(a)]
Gravel side slope on surface barrier	3.0	33

np = Not provided.
(a) Inferred from estimate for Warden silt loam.

3.0 Affected Environment

An adequate evaluation of recharge on the Central Plateau, and specifically in and around tank farms on the Plateau, requires an understanding of the local environment. This section summarizes information on the climate and meteorology, geology and soil, hydrology, and ecology. Portions of this section were extracted from existing reports, including Neitzel et al. (2005), Hoitink et al. (2005), Reidel and Chamness (2007), Fayer and Szecsody (2004), Hartman et al. (2007), and Horton (2007). For brevity, references in the original texts are not included here.

The Hanford Site lies within the semiarid Pasco Basin of the Columbia Plateau in southeastern Washington State (Figure 1.1). The Hanford Site occupies an area of about 1,517 km^2. The Columbia River flows through the northern part of the Hanford Site and forms part of the Site's eastern boundary. The Yakima River runs near the southern boundary of the Hanford Site and joins the Columbia River at the city of Richland, which bounds the Hanford Site on the southeast. Rattlesnake Mountain, Yakima Ridge, and Umtanum Ridge form the southwestern and western boundaries. The Saddle Mountains form the northern boundary of the Hanford Site. Two small east-west ridges, Gable Butte and Gable Mountain, rise above the plateau of the central part of the Hanford Site. Adjoining lands to the west, north, and east are principally range and agricultural land. The cities of Kennewick, Pasco, and Richland (Tri-Cities) constitute the nearest population centers and are located south-southeast of the Hanford Site.

3.1 Climate and Meteorology

The Cascade Mountains, 100 km to the west, greatly influence the climate of the Hanford area by means of their rain-shadow effect. This mountain range also serves as a source of cold air drainage, which has a considerable effect on the wind regime on the Hanford Site. Climatological data have been collected at the Hanford Meteorological Station (HMS) since 1945 (Hoitink et al. 2005). The HMS is sited between the 200 East and 200 West Areas at an elevation of 223 m. The data are representative of the general climatic conditions for the region and describe the specific climate of the Central Plateau.

3.1.1 Precipitation

Between 1946 and 2004, annual precipitation at the HMS averaged 173 mm and varied between 76 and 313 mm. Table 3.1 shows how monthly averages have varied in that time. The wettest season on record was the winter period from December 1996 to February 1997, with 138 mm of precipitation; the driest season was the summer of 1973 (June–August) when only 1 mm of precipitation was measured. On average, half of the annual precipitation occurs during the four months from November through February. A rainfall intensity of 20 mm/hr persisting for 1 hour is expected only once every 1,000 years. A day with 18 mm precipitation is expected to occur about once every 2 years, while a day with 54 mm precipitation is expected only once every 1,000 years. Hanford nearly experienced such a 1,000-yr event when it received 48.5 mm in a 24-hr period in October 1957.

Snowfall accounts for about 42% of all precipitation from November through February. Monthly average snowfall is greatest in December (130 mm) and January (127 mm). The record monthly snowfall of 594 mm occurred in January 1950. The seasonal record snowfall of 1,425 mm occurred during the

Table 3.1. Monthly Precipitation Variations Between 1946 and 2004 at the Hanford Meteorological Station

Month	Monthly Precipitation (mm)		
	Maximum	Mean	Minimum
January	62.7	24.1	2.0
February	53.3	16.4	0.0
March	47.2	12.7	0.5
April	56.6	12.0	0.0
May	51.6	13.0	0.0
June	74.2	13.5	0.0
July	44.7	5.3	0.0
August	34.5	6.3	0.0
September	34.0	7.7	0.0
October	69.1	13.6	0.0
November	67.8	22.3	0.0
December	93.7	25.7	1.8
Annual	313	173	75.9

winter of 1992–1993. This amount has a return period of almost 500 years. On average, the first measurable snow appears on November 30 and the last on February 13. Since 1946, snow has been measured as early as October 26 and as late as April 30.

3.1.2 Air Temperature

Table 3.2 shows the range of monthly temperatures since 1946. The highest winter monthly average temperature was 6.9°C in February 1958 and 1991, while the lowest average temperature was −11.1°C in January 1950. The highest summer monthly average temperature was 27.9°C in July 1985, while the lowest average temperature was 17.2°C in June 1953. There were, on average, 53 days during the months of April through September with maximum temperatures ≥32°C and 12 days with maxima ≥38°C. During the months of October through March, an average of 106 days had temperature minimums below 0°C; an average of 2 days had minimum temperatures that were ≤−18°C. The record maximum temperature is 45°C; the record minimum temperature is −31°C. The potential for plant activity can be represented by the number of growing days, which is the number of days between the last freezing temperature in spring and the first freezing temperature in autumn. Since 1945, the number of growing days has averaged 181 days per year, with annual values ranging from 142 (in 1974) and 216 days (in 1994).

3.1.3 Humidity

Since 1950, the average annual relative humidity at the HMS has been 55%; annual values ranged from 50 to 59%. December had the highest monthly average humidity (80%), with values that ranged from 69 to 91%. July had the lowest monthly average humidity (33%), with values that ranged from 22 to 46%.

Table 3.2. Monthly Air Temperature Variations Between 1946 and 2004 at the Hanford Meteorological Station

Month	Monthly Air Temperature (°C)		
	Maximum	Mean	Minimum
January	5.8	−0.5	−11.1
February	6.9	3.2	−3.6
March	10.8	7.4	4.1
April	14.6	11.6	8.6
May	20.4	16.6	13.3
June	24.9	20.8	17.2
July	27.9	24.7	21.4
August	27.5	24.0	21.0
September	22.4	19.0	14.9
October	15.3	11.7	8.8
November	8.1	4.5	−4.0
December	3.6	0.3	−6.1
Annual		11.9	

3.1.4 Solar Radiation

Since 1953, the average annual daily solar radiation at the HMS has been 171 W/m^2 (352 langleys). Average daily values were lowest in December (40 W/m^2) and highest in July (304 W/m^2). The lowest observed daily value was 2.9 W/m^2 in December 2002; the highest observed daily value was 406 W/m^2 in May 1977.

3.1.5 Wind

Prevailing wind directions on the Central Plateau were either from the west-northwest or northwest in all months of the year. At a height of 15.2 m, average annual wind speed was 12.2 km/hr. Monthly average wind speeds were lowest during the winter months (9.7 km/h in December) and highest during the summer (14.6 km/hr in June). Peak wind gusts in every month originated from the west-southwest, southwest, and south-southwest; monthly peak gusts varied from 105 to 129 km/hr.

3.2 Geology

Reidel and Chamness (2007) provide an extensive review of the geology of the Hanford Site as it relates to the SST WMAs. Elements of their review as well as material from Neitzel (2005) are included here to provide the geologic context for understanding recharge.

The Hanford Site lies within the Columbia Plateau, which was formed between 17 and 6 million years ago, mainly from a thick sequence of basalt flows known as the Columbia River Basalt Group (CRBG). These flows have been folded and faulted over the past 17 million years, creating broad structural and topographic basins separated by asymmetric anticlinal ridges. The Hanford Site lies within one of the larger basins, the Pasco Basin. The Pasco Basin is bounded on the north by the Saddle Mountains and on the south by Rattlesnake Mountain and the Rattlesnake Hills. Yakima Ridge and Umtanum Ridge trend into the basin and subdivide it into a series of smaller anticlinal ridges and synclinal basins. The largest syncline, the Cold Creek syncline, lies between Umtanum Ridge and Yakima Ridge and is the principal structure containing the SST WMAs.

Most post-CRBG sediments are confined to the synclinal valleys. The dominant source of sediment between the upper Miocene to middle Pliocene (10 to 3 million years ago) was the Columbia River system. The upper Ellensburg Formation and the Ringold Formation are the main sediment elements formed during that period. Capping the sedimentary sequence in the synclines and basins are sediments comprising the Pleistocene Hanford formation deposited during cataclysmic floods and recent eolian deposits.

Cataclysmic floods inundated the Pasco Basin several times during the Pleistocene epoch when ice dams failed in northern Washington and Idaho. As the floods raced across the lowlands of the Pasco Basin and Hanford Site, the floodwaters lost energy and left behind deposits of gravel, sand, and silt. The Cold Creek bar is a geomorphic remnant of those cataclysmic floods. A sizable portion of the bar is elevated above the surrounding terrain by 15.2 to 30.5 m (50 to 100 ft), forming the Central Plateau; the SST WMAs are situated on the plateau.

The stratigraphy of the WMAs consists of the basalt flows overlain by the Ringold Formation, the Hanford formation, and Holocene eolian deposits. All recharge-related measurements and estimates occur within the Hanford formation and eolian deposits, which are described in the following paragraphs.

3.2.1 Hanford Formation

The Hanford formation is the main stratigraphic unit at the surface of the tank farms. According to Reidel and Chamness (2007), the thickness of the Hanford formation ranges from about 30.5 m (100 ft) in the 200 West Area to as much as 113 m (370 ft) in the 200 East Area.

The Hanford formation consists of pebble to boulder gravel, fine- to coarse-grained sand, and silt. These deposits are divided into three facies: 1) gravel-dominated, 2) sand-dominated, and 3) sand- and silt-dominated. These facies are referred to as coarse-grained deposits, plane-laminated sand facies, and rhythmite facies, respectively.

The gravel-dominated facies association generally consists of coarse-grained basaltic sand and granule to boulder gravel. These deposits display massive bedding, plane to low-angle bedding, and large-scale planar cross-bedding in outcrop. The gravel facies dominates the Hanford formation in the northern part of the 200 East and West Areas. The gravel-dominated facies was deposited by high-energy floodwaters in or immediately adjacent to the main cataclysmic flood channels.

The sand-dominated facies association consists of fine- to coarse-grained sand and granule gravel displaying plane lamination and bedding and, less commonly, plane bedding and channel-fill sequences in outcrop. These sands may contain small pebbles and rip-up clasts in addition to pebble-gravel interbeds and silty interbeds less than 1 m thick. The silt content of these sands is variable. These sands typically are basaltic, commonly being referred to as black, gray, or salt-and-pepper sands. This facies is most common in the central Cold Creek syncline, in the central to southern parts of the 200 East and 200 West

Areas. The laminated sand facies was deposited adjacent to main flood channels during the waning stages of flooding. The laminated sand facies is transitional between the gravel-dominated facies and the rhythmite facies.

The interbedded sand- and silt-dominated facies association consists of thinly bedded, plane-laminated and ripple cross-laminated silt and fine- to coarse-grained sand that commonly display normally graded rhythmites a few centimeters to several tens of centimeters thick. This facies is found throughout the central, southern, and western Cold Creek syncline within and south of the 200 East and 200 West Areas. These sediments were deposited under slack water conditions and in back-flooded areas.

At many locations on the Hanford Site, variably oriented sediment features known as clastic dikes cut across the typically horizontal sediment layers (Fecht et al. 1999). For years, clastic dikes were thought to be preferential pathways for downward water and contaminant transport as well as hindrances to lateral spreading of water and contaminants such that downward flow is enhanced. A recent study by Murray et al. (2007) involving field experiments and numerical simulation supports the hypothesis that clastic dikes can be fast pathways for water movement depending on the flow conditions and degree of heterogeneity within the specific clastic dike. Murray et al. (2007) also observed that vertical transport of contaminants might be enhanced for non-reactive contaminants like tritium and technetium-99, but that more reactive contaminants would be unlikely to travel far in clastic dikes because of the presence of significant quantities of fine-grained particles (up to 19% clay sized material in the study by Murray et al. 2007) with which the contaminants could react.

3.2.2 Holocene Deposits

The Holocene Epoch (i.e., the last 10,000 years) was characterized by wind mobilization and deposition of sediments. Holocene deposits consisting of silt, sand, and gravel form a thin veneer across the Central Plateau. The thickness of the eolian material ranges from nearly zero in highly disturbed areas such as the WMAs (Reidel and Chamness 2007) to as much as 1.5 m in the southern part of the 200 East Area (Fayer and Szecsody 2004). In the special case of sand dunes (e.g., the sand dunes south of the 200 Areas), the thickness exceeds 1.5 m.

3.3 Soils

The Holocene deposits and exposed Hanford formation sediment have experienced soil development and evolved into identifiable soil types. Hajek (1966) produced a soil map of the Hanford Site. Figure 3.1 shows that five soil types cover the bulk of the Central Plateau—Rupert sand (also known as Quincy sand), Burbank loamy sand, Ephrata sandy loam, Esquatzel silt loam, and Hezel sand. Hajek (1966) described these types of soil as follows (verbatim from Hajek 1966):

> ***Rupert Sand***. This mapping unit represents one of the most extensive soils on the Hanford Project. The surface is a brown to grayish brown (10YR5/2[a]) coarse sand, which grades to a dark grayish brown (10YR4/2) sand at about 36 in. Rupert soils developed under grass, sagebrush, and hopsage in coarse sandy alluvial deposits, which were mantled by wind-blown

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(a) Character sets like this – 10YR5/2 – represent a code that identifies soil color using the Munsell color system.

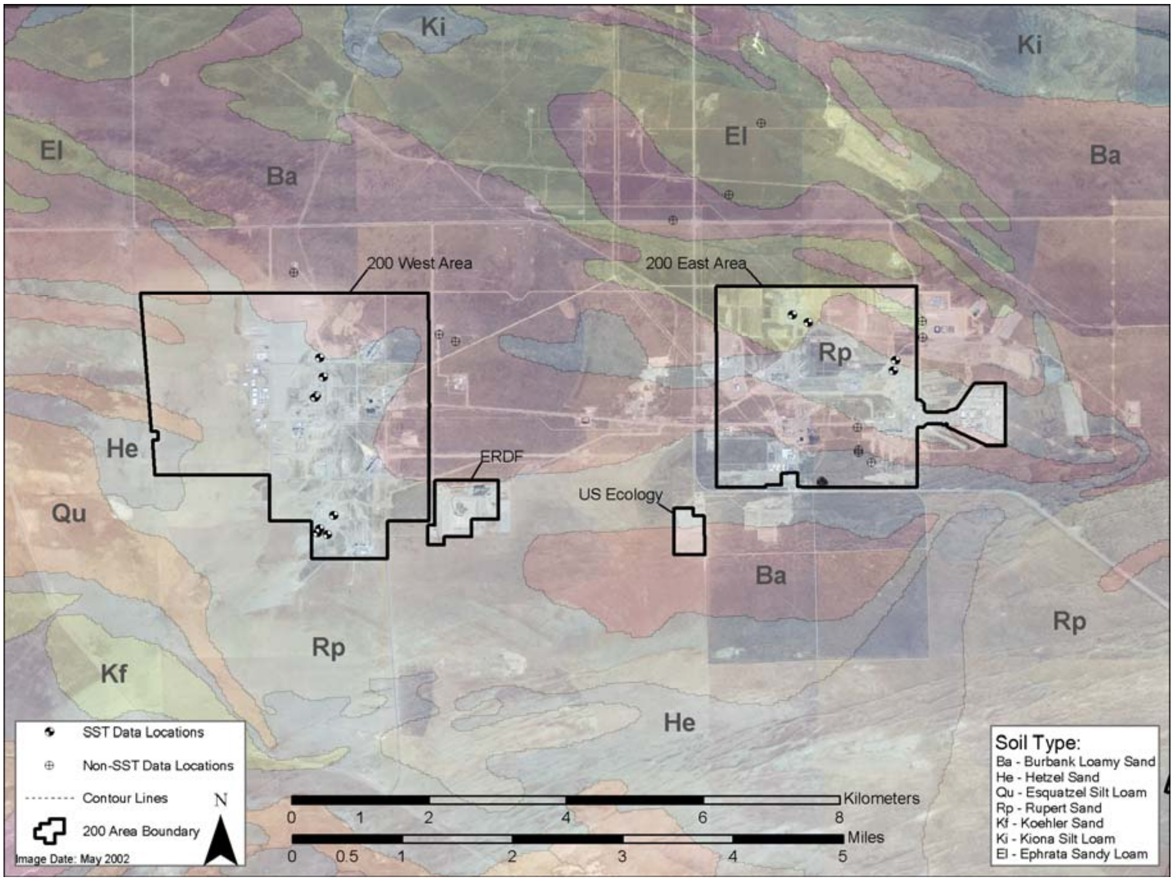

Figure 3.1. Soil Types on the Central Plateau

sand. Relief characteristically consists of hummocky terraces and dune-like ridges. This soil may be correlated as Quincy sand, which was not separated here. Active sand dunes are present. Some dune areas are separated; however, many small dunes, blow-outs, and associated small areas of Ephrata and Burbank soils are included.

Burbank Loamy Sand. This is a dark-colored [surface is very dark grayish brown (10YR3/2); subsoil is dark grayish brown (10YR4/2)], coarse-textured soil which is underlain by gravel. The surface soil is usually about 16 in. thick but can be 30 in. thick. The gravel content of the subsoil may range from 20 to 80 volume percent. Areas of Ephrata and Rupert are included.

Ephrata Sandy Loam. This is a dark colored surface is very dark grayish brown (10YR3/2); subsoil is dark grayish brown (10YR4/2) medium-textured soil which is underlain by gravelly material which may continue for many feet. This soil is associated with the Burbank soil and many small areas were included in delineations of this soil type. The topography is generally level.

Esquatzel Silt Loam. This is a deep dark brown (10YR3/3) soil formed in recent alluvium derived from loess and lake sediments. The subsoil grades to dark grayish brown (10YR4/2) in many areas but color and texture of the subsoil is variable due to the stratified nature of the alluvial deposits. Esquatzel soils are associated with Ritzville and Warden and often seem to have developed from sediments eroded from these two series.

Hezel Sand. Hezel soils are similar to Rupert sands; however, a laminated grayish brown (10YR5/2) strongly calcareous silt loam subsoil is usually encountered within 40 in. of the surface. The surface soil is very dark brown (10YR3/3) and was formed in wind-blown sands which mantled lake-laid sediments. Areas of Rupert, Burbank, and blow-outs are included.

The soil map produced by Hajek (1966) was based largely on the soil survey work conducted around 1910 to 1915 and reported by Kocher and Strahorn (1919). The focus of those surveys was primarily agricultural use and not estimation of natural recharge. Recent excavations suggest that the soil conditions around the 200 Areas are unique and not easily classified into the above four soil types. For example, Figure 3.2 shows the side of a pit that was excavated about 175 m west of the IDF site in the 200 East Area. The profile shows a 1.2-m-thick set of nearly horizontal layers of alternating sands, gravels, and fines that were deposited during the waning period of the last cataclysmic flooding. Above this sequence of layers lies eolian material; below this sequence lies the coarse sand of the Hanford formation.

Anecdotal information suggests that some variation of the sequence of layers in Figure 3.2 exists across much of the Central Plateau. The contrasting textures within that sequence create capillary breaks and thin fine-textured layers that impede the downward movement of liquid water. The water storage capacity of the eolian material residing above the layers will influence the potential deep drainage rate. Depths of eolian material range from zero (where it has been removed during construction; see Reidel and Chamness 2007) to more than 1.5 m (Fayer and Szecsody 2004). Depths between 1.0 and 2.0 m may be ideal for storing all precipitation until it can be removed by evapotranspiration, thus significantly reducing deep drainage rates. If thinner than 1.0 m, the eolian material may not be able to store all winter precipitation. If thicker than 2.0 m, the eolian material can store the precipitation, but the water stored near the deep capillary break may be too deep to be removed completely by evapotranspiration. In either case, the result is an increased potential for higher drainage rates.

3.4 Topography

Figure 3.3 shows how topography varies across the Central Plateau. The highest point (240 m) occurs in the northwestern corner of the 200 West Area and the lowest (180 m) in the northeastern corner of the 200 East Area. Single-shell tank WMAs are at elevations between 195 and 215 m.

3.5 Hydrology

The primary surface-water features associated with the Hanford Site are the Columbia and Yakima rivers. The Columbia River is the second largest river in the contiguous United States in terms of total flow and is the dominant surface-water body on the Hanford Site. Cold Creek and its tributary, Dry Creek, are ephemeral (i.e., not always flowing) streams within the Yakima River drainage system. Both streams drain areas along the western part of the Hanford Site. Surface flow, which may occur during spring runoff or after heavier-than-normal precipitation, infiltrates and disappears into the surface sediments. The Central Plateau site is well above and away from these surface-water features and is unaffected by them in any direct manner. However, a probable maximum flood analysis suggested the southwestern corner of the 200 West Area would be inundated (Skaggs and Walters 1981). The discharge rate of such an event would be four times larger than the 100-year flood. Neither event has occurred in historical times.

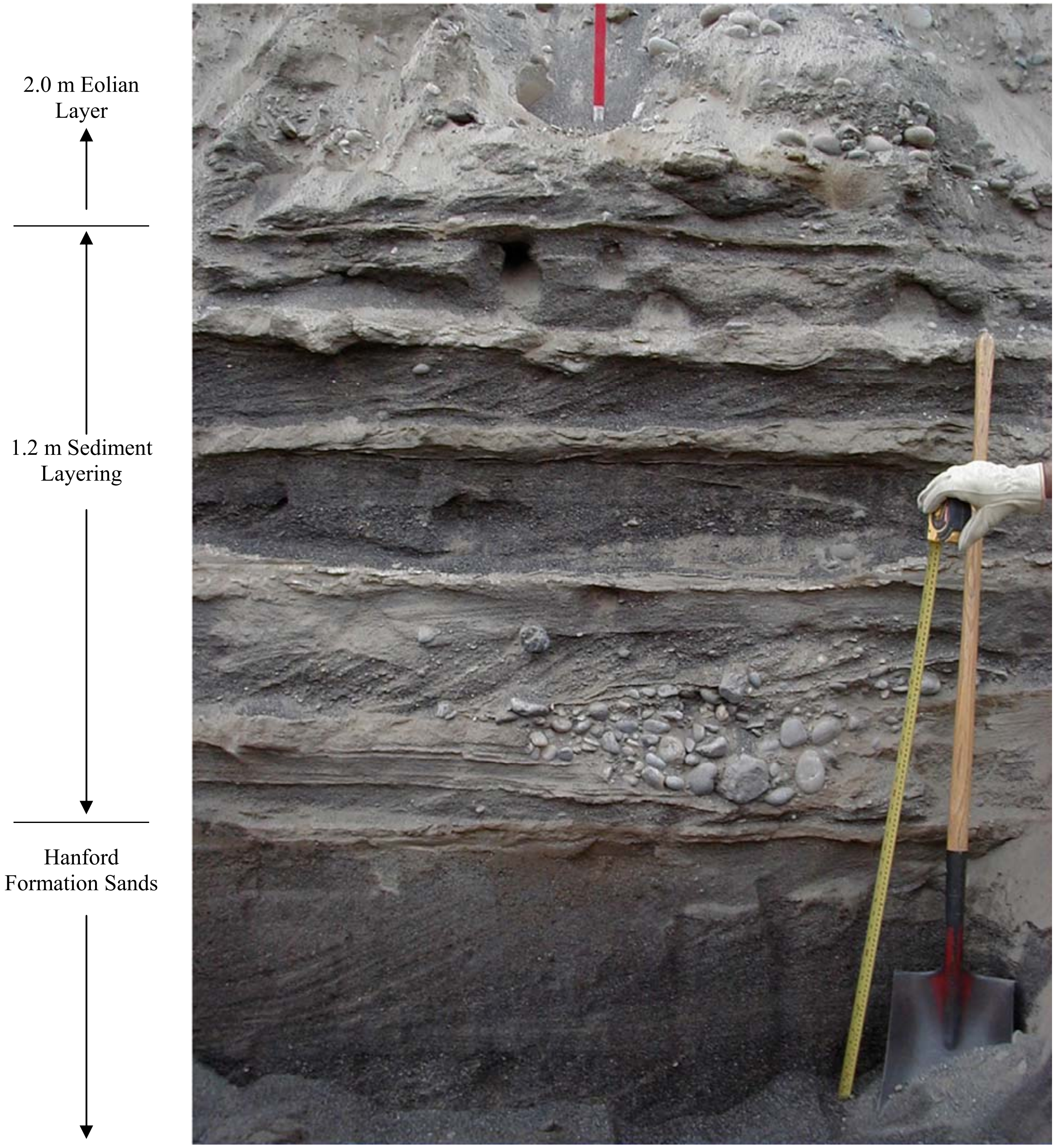

Figure 3.2. Sediment Layering in a Pit Excavated 175 m West of the Southwestern Corner of the Integrated Disposal Facility. The pronounced sediment layering is the result of multiple flood events that deposited alternating layers varying from fine sand to coarse sand and gravel. (Photo courtesy of Dr. John Selker, Oregon State University; after Fayer and Szecsody 2004).

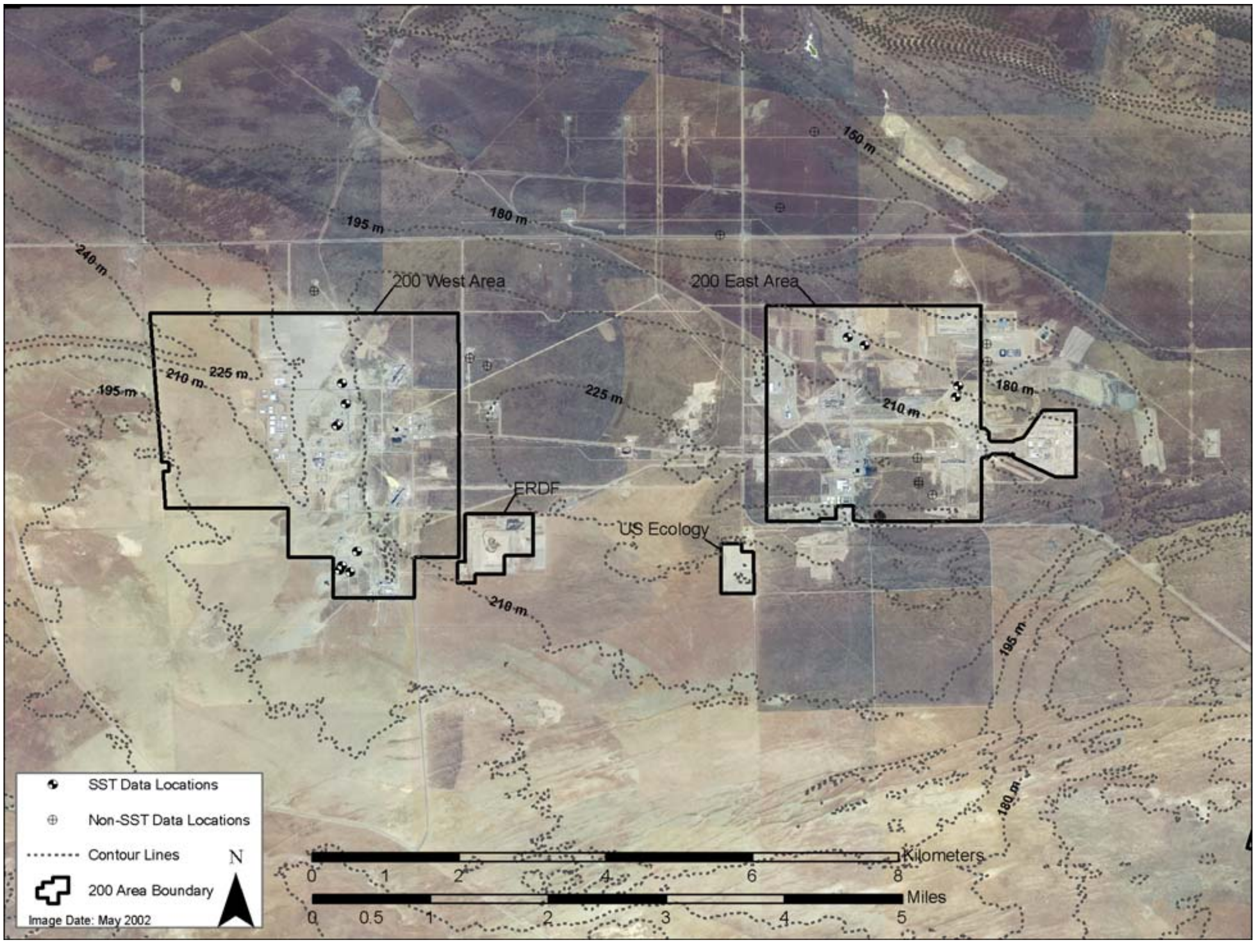

Figure 3.3. Topography of the Central Plateau

Locally, overland flow has not been considered an important process in natural settings because of the lack of erosion features that might document past overland flow events. Direct observations of overland flow are rare, and they typically occur when there is rapid snowmelt and frozen soil. During such events, erosion is not likely; thus, an erosion signature of the event would not be left behind. In operational areas of the Site, overland flow has been observed more frequently and is discussed in greater detail in Section 4.

Average annual natural recharge rates across the Hanford Site range from near 0 to around 110 mm/yr, depending on surface conditions (Gee et al. 1992; Fayer and Keller 2007). Low recharge rates occur in fine-textured sediments where deep-rooted plants occur. The larger values occur in areas having a coarse gravelly surface and no vegetative cover (e.g., disturbed areas such as around the tank farms).

The vadose zone (also called the unsaturated zone) beneath the land surface of the Central Plateau ranges from about 65 m thick in the 200 West Area to 92 m thick in the 200 East Area. This vadose zone consists primarily of Hanford formation sediments capped with Holocene eolian sediment. In some locations, Ringold Formation sediments occupy the lower vadose zone.

The Pasco Basin has several confined aquifers within the basalt flows and one unconfined aquifer above the basalt flows. In the 200 Areas, the aquifer above the basalt is unconfined to locally semi-confined and is contained largely within the sediments of the Ringold Formation and Hanford formation. In some locations, the aquifer does not exist above the basalt.

The water table beneath the Central Plateau occurs within either the Hanford formation or Ringold Formation, depending on location, or is absent where basalt rises above the plane of the water table surface. Generally, groundwater flows from the western upland areas to the east and eventually north or southeast. However, artificial recharge from wastewater disposal activities has perturbed the flow directions. In the 200 East Area, the water table is so flat that local flow direction is difficult to discern.

3.6 Ecology

The Central Plateau portion of the Hanford Site is characterized as a shrub-steppe ecosystem that is adapted to the region's mid-latitude semiarid climate of hot summers, cold winters, and low precipitation (Neitzel 2005). Such ecosystems are typified by a shrub overstory (the high elements in the landscape) with a grass and forb understory (the low elements in the landscape). Lichens and mosses, often referred to as *microbiotic or cryptogamic crust*, provide a soil-stabilizing growth on undisturbed soils in the shrub-steppe ecosystem.

In the early 1800s, dominant plants in the area were big sagebrush (*Artemisia tridentata*) and an understory consisting of perennial Sandberg's bluegrass (*Poa sandbergii*) and bluebunch wheatgrass (*Pseudoregneria spicata*). Other species included three-tip sagebrush (*Artemisia tripartite*), bitterbrush (*Purshia tridentata*), gray rabbitbrush (*Ericameria nauseosa*), spiny hopsage (*Grayia spinosa*), needle-and-thread grass (*Hesperostipa comata*), Indian ricegrass (*Achnatherum hymenoides*), and prairie Junegrass (*Koeleria cristata*).

With the advent of settlement, livestock grazing and agricultural production contributed to colonization by non-native vegetation species that currently dominate portions of the landscape. Although agriculture and livestock production were the primary subsistence activities at the turn of the century, these activities ceased when the Hanford Site was established in 1943. Range fires that historically burned through the area during the dry summers eliminate fire-intolerant species (e.g., big sagebrush) and allow more opportunistic and fire-resistant species to establish. Of the 727 species of vascular plants recorded for the Hanford Site, approximately 25% are non-native. The dominant non-native species, cheatgrass (*Bromus tectorum*), is an aggressive colonizer and has become well established across the Site.

Today, the undisturbed portions of the Central Plateau are characterized as sagebrush/cheatgrass or sagebrush/Sandberg's bluegrass communities. In contrast, the waste management areas on the Central Plateau are generally either barren or vegetated by nonnative species. Those that are barren reflect poor soil conditions created by waste management activities or active attempts to keep sites free of vegetation, using herbicides, to prevent plant root intrusion to waste constituents. Those waste sites that are vegetated are so because they were planted with crested or Siberian wheatgrass to stabilize surface soil, control soil moisture, or displace more invasive deep-rooted species like Russian thistle. Alternatively, if not actively planted, these sites have experienced invasion by both native and non-native species.

Approximately 300 species of terrestrial vertebrates have been observed on the Hanford Site, including approximately 46 species of mammals, 246 species of birds, 6 species of amphibians, and

10 species of reptiles. Terrestrial wildlife include Rocky Mountain elk (*Cervus elaphus*), mule deer (*Odocoileus hemionus*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), badger (*Taxidea taxus*), deer mice (*Peromyscus maniculatus*), harvest mice (*Riethrodontonomys megalotis*), ground squirrels (*Spermophilus washingtonii*), voles (*Microtus montanus*), and black-tailed jackrabbits (*Lepus californicus*). The most abundant mammal on the Hanford Site is the Great Basin pocket mouse (*Perognathus parvus*). Bird species commonly found in the shrub-steppe habitats at Hanford include the western meadowlark (*Sturnella neglecta*), horned lark (*Eremophila alpestris*), long-billed curlew (*Numenius americanus*), vesper sparrow (*Pooecetes gramineus*), sage sparrow (*Amphispiza belli*), sage thrasher (*Oreoscoptes montanus*), loggerhead shrike (*Lanius ludovicianus*), and burrowing owls (*Athene cunicularia*).

Butterflies, grasshoppers, and darkling beetles are among the more conspicuous of the approximately 1,500 species of insects that have been identified from specimens collected on the Hanford Site. The actual number of insect species occurring on the Hanford Site may reach as high as 15,500.

The side-blotched lizard (*Uta stansburiana*) is the most abundant reptile species that occurs on the Hanford Site. Short-horned (*Phrynosoma douglassii*) and sagebrush (*Sceloporus graciosus*) lizards are reported but occur infrequently. The most common snake species include gopher snake (*Piteriphis melanoleucus*), yellow-bellied racer (*Coluber constrictor*), and Western rattlesnake (*Crotalus viridis*).

The above summary of Hanford ecology (from Neitzel 2005) is based on data collected across the entire Hanford Site. In addition to the common species identified by Neitzel (2005), some species may be important not for their numbers but for their potential impacts to waste sites. For example, harvester ants (*Pogonomyrmex* spp.) have been observed at Hanford and can burrow and bring waste material to the surface.

4.0 Single-Shell Tank Waste Management Area Information

Estimating recharge in and around the tank farms requires more specific information about the WMAs than is provided in Section 3. This section provides that information by describing the construction of the farms, reviewing the environment-specific features of the WMAs, and showing aerial views of each WMA as seen in 2002. More details can be found in Chapter 3 and in the numerous FIR documents (e.g., CH2M Hill 2002; Knepp 2002; Myers 2005)

4.1 Tank Farm Construction

Construction of SST farms occurred in stages. Four farms were started in 1943 and completed by 1944. Figure 4.1 shows the typical construction conditions, including the large-scale disturbance in and around the farms that resulted from construction activities. The rest of the SST farms were started and finished at various times between 1946 and 1964. Distributed within each farm are up to four different tank types defined by capacity: 55, 530, 750, and 1,000 kgal. These tank sizes represent 11, 40, 32, and 17%, respectively, of the 149 SSTs. On a total volume basis, the tank sizes represent 1, 36, 41, and 22%, respectively, of the total volume of SST waste. Table 4.1 summarizes the specific construction dates and tank types for each farm.

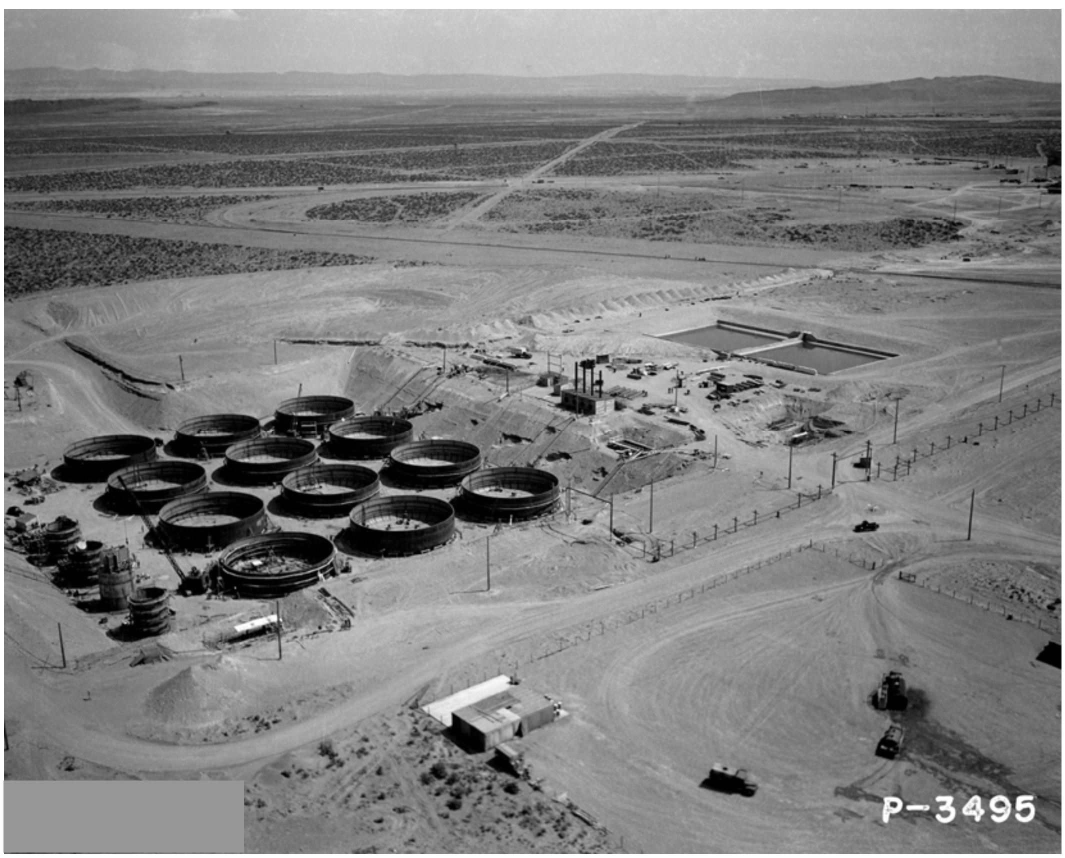

Figure 4.1. The SST WMA T Under Construction in 1944. Each large tank is 22.8 m (75 ft) in diameter, and the base of the excavation is roughly 13.7 m (45 ft) below ground surface. View is to the northeast toward Gable Mountain in the distance.

Table 4.1. Single-Shell Tank Waste Management Area Construction Dates and Tank Sizes

Waste Management Area	Tank Farm	Construction Years	Years from Completion to 2007	Number of Tank Types[(a)]			
				Type I	Type II	Type III	Type IV
A-AX	A	1954–1955	52				6
	AX	1963–1964	43				4
B-BX-BY	B	1943–1944	63	4	12		
	BX	1946–1947	60		12		
	BY	1948–1949	58			12	
C	C	1943–1944	63	4	12		
S-SX	S	1950–1951	56			12	
	SX	1953–1954	53				15
T	T	1943–1944	63	4	12		
TX-TY	TX	1947–1948	59			18	
	TY	1951–1952	55			6	
U	U	1943–1944	63	4	12		
Total Tanks/Type				**16**	**60**	**48**	**25**

(a) Tank sizes are as follows:
- Type I: 200 Series, 208 m^3 (55,000 gal)
- Type II: 100 Series, 2,006 m^3 (530,000 gal)
- Type III: 100 Series, 2,839 m^3 (750,000 gal)
- Type IV: 100 Series, 3,785 m^3 (1,000,000 gal).

Given the construction completion dates, we calculated the length of time that the tanks have been in place; Table 4.1 shows that those times vary between 43 and 63 years. The length of service is important in understanding the evolution of the water content, water potential, and tracer profiles within the backfill material.

Table 4.2 contains nominal dimensions for each tank type in each farm. Depth of the concrete top of the tanks below ground surface can be used to evaluate the impact on recharge rates. Tank height and width can be used to determine the volume of the vadose zone occupied by the tanks and evaluate whether their presence affects recharge conditions. The outer width of the concrete footings is provided to indicate the extent to which recharge above the tank must be diverted before it can proceed downward to the water table. Depth of water table below the tank bottom is provided for cases when an estimate of travel time to the water table is needed.

Section 2 explained that buried tanks do not enhance recharge. Instead, they redirect vadose zone water laterally, which creates much higher fluxes along the sides of the tanks than would normally occur. At the water table, there are likely to be localized zones of higher and lower fluxes (e.g., beneath tank edges versus tank centers; preferential flow paths), but the overall average recharge for the tank farm should remain the same (as if the tanks were not there). From a contaminant transport perspective, however, the altered vadose zone fluxes are extremely important and their magnitude and variability need to be addressed in the RCRA Facility Investigation.

The increase in flux along the widest point of a tank, i.e., the footing at the base, was approximated to provide some perspective. We assumed that all recharge water that reaches the tank is diverted and flows downward and within 1 m of the outside edge of the tank base footing. Table 4.2 shows that the largest

Table 4.2. Single-Shell Tank Dimensions

Waste Management Area	Tank Farm	Tank Types[a]	Nominal Dimensions (m)			
			Depth Below Surface of Tank Top Concrete	Tank Height (top of concrete to bottom of concrete)	Radius of Outer Width of Concrete Footings	Water Table Depth Below Tank Bottom (April 2006)
A-AX	A	IV	2.3	14.3	13.0	72
	AX	IV	2.3	14.3	13.3	69
B-BX-BY	B	I	1.8	6.1	3.7	65
	B	II	1.8	8.5	12.4	65
	BX	II	2.7	9.8	12.4	67
	BY	III	2.4	11.9	12.7	62
C	C	I	1.8	6.1	3.7	66
	C	II	2.7	8.5	12.4	66
S-SX	S	III	2.4	11.9	12.4	54
	SX	IV	2.3	14.3	12.6	51
T	T	I	1.8	6.1	3.7	60
	T	II	2.7	8.5	12.4	60
TX-TY	TX	III	2.4	11.9	12.7	55
	TY	III	2.4	11.9	13.3	52
U	U	I	1.8	6.1	3.7	56
	U	II	2.7	8.5	12.4	56

(a) Nominal tank sizes are as follows:

- Type I: 200 Series, 208 m^3 (55,000 gal)
- Type II: 100 Series, 2,006 m^3 (530,000 gal)
- Type III: 100 Series, 2,839 m^3 (750,000 gal)
- Type IV: 100 Series, 3,785 m^3 (1,000,000 gal).

radius is 13.3 m. Using a nominal recharge rate of 100 mm/yr, the increase in flux moving past the footing is 665 mm/yr, which when added to the existing recharge flux of 100 mm/yr yields a total flux of 765 mm/yr. In effect, the tank has increased the vadose zone flux moving past the footing by a factor of 7.65. It is entirely possible that the additional flux will spread out farther than 1 m from the footing. The closest spacing between footings of any two tanks is roughly 5.77 m. In this extreme case, accounting for spreading out to half that distance (2.88 m) yields a total flux of 331 mm/yr (flux increase of 231 mm/yr plus recharge flux). Using the backfill hydraulic properties reported by CH2M HILL Hanford (2002), we estimated the changes that might be observed. If spreading were out to the midpoint between tanks, the water content would increase by about 1 cm^3/cm^3 and suction head would decrease from 238 to 152 cm.

4.2 Single-Shell Tank Environmental Conditions

Much of the discussion of environmental conditions in Section 3 is relevant to SST conditions. However, there are specific features of the SST farms that warrant additional discussion. These features touch on weather, soil, topography, hydrology, and ecology.

4.2.1 Weather

Although the HMS is the primary weather station at the Hanford Site, more than 20 satellite weather stations are maintained across the Site, some dating back to the 1990s. One is located in the 200 East Area and one in the 200 West Area. Hoitink et al. (2005) and predecessor reports show some variability in mean monthly and annual air temperatures at these two locations, but the general trends are very consistent. Of note is that the air temperature in the 200 East Area is, on average, 1.2 degrees warmer than at the HMS. This could be explained easily by the different surface conditions surrounding the stations. The HMS is in the midst of mature shrub-steppe habitat. The satellite stations are in the midst of roads, buildings, and graveled lots.

Precipitation shows monthly and annual variability across the Plateau. On average, annual amounts at the 200 East and West stations are 2 in. less than at the HMS. The differences may be the result of not having constant attention from a meteorologist, particularly during the measurement of winter precipitation. There have been infrequent occurrences of thunderstorms that are very local. Ward et al. (2005) described one example in Spring 2004, when a major thundershower fell in the vicinity of the BX-BY Tank Farm, yet was barely detected at the HMS.

4.2.2 Soils

Construction of tank farms involved major excavation down to 13.7 m (45 ft), stockpiling of sediment, and backfilling around the tanks (e.g., see Figure 4.1). The type of near-surface layering seen in Figure 3.2 is no longer present in the tank farms. What exists in and around the tank farms today are unique soil types that have not yet been characterized except in the general sense of being a mix of Hanford formation sands and gravels. Although it has long been assumed that the tank farm surfaces are highly porous, events such as ponding (Topography section) and runoff (Hydrology section) suggest that the assumption is not universally met.

4.2.3 Topography

Figures 4.2 and 4.3 show detailed topography for the 200 West and East Areas, respectively. All but two of the farms reside on relatively flat terrain. The lack of any significant slope can be disadvantageous if (for whatever reason) water accumulates on the surface. Such an event happened in February 1979 when rapid snowmelt over frozen soil resulted in water ponding on the surface with no clear path to flow off the farm (Hodges 1998), and Figure 4.4 shows the outcome. Hodges (1998) observed that both WMAs TX-TY and T were sited at the bottom of a natural depression. In the case of T farm, runoff from slightly higher ground along the east side flowed onto T farm but had no outlet. Hodges (1998) noted "There is also little doubt that this is not the only event of this type that has occurred." One of the recommendations made by Hodges (1998) was to take corrective measures to reduce infiltration from natural causes such as surface runoff and ponding of snowmelt. The 1979 overland flow event at T farm as well as other events elsewhere eventually led to a concerted effort to establish runon/runoff controls throughout the SST WMAs (e.g., Parsons 1996; CH2M HILL 2002). Wood and Jones (2003) attempted to correlate snowmelt and maximum monthly precipitation with groundwater contamination occurrences at WMA U.

The two farms with significant topography change are BY and C. Although the surfaces directly above the tanks in these farms are relatively flat, the perimeters of the farms have steep side slopes that are heavily armored with gravel. Runoff from BY farm is described in Section 4.2.4.

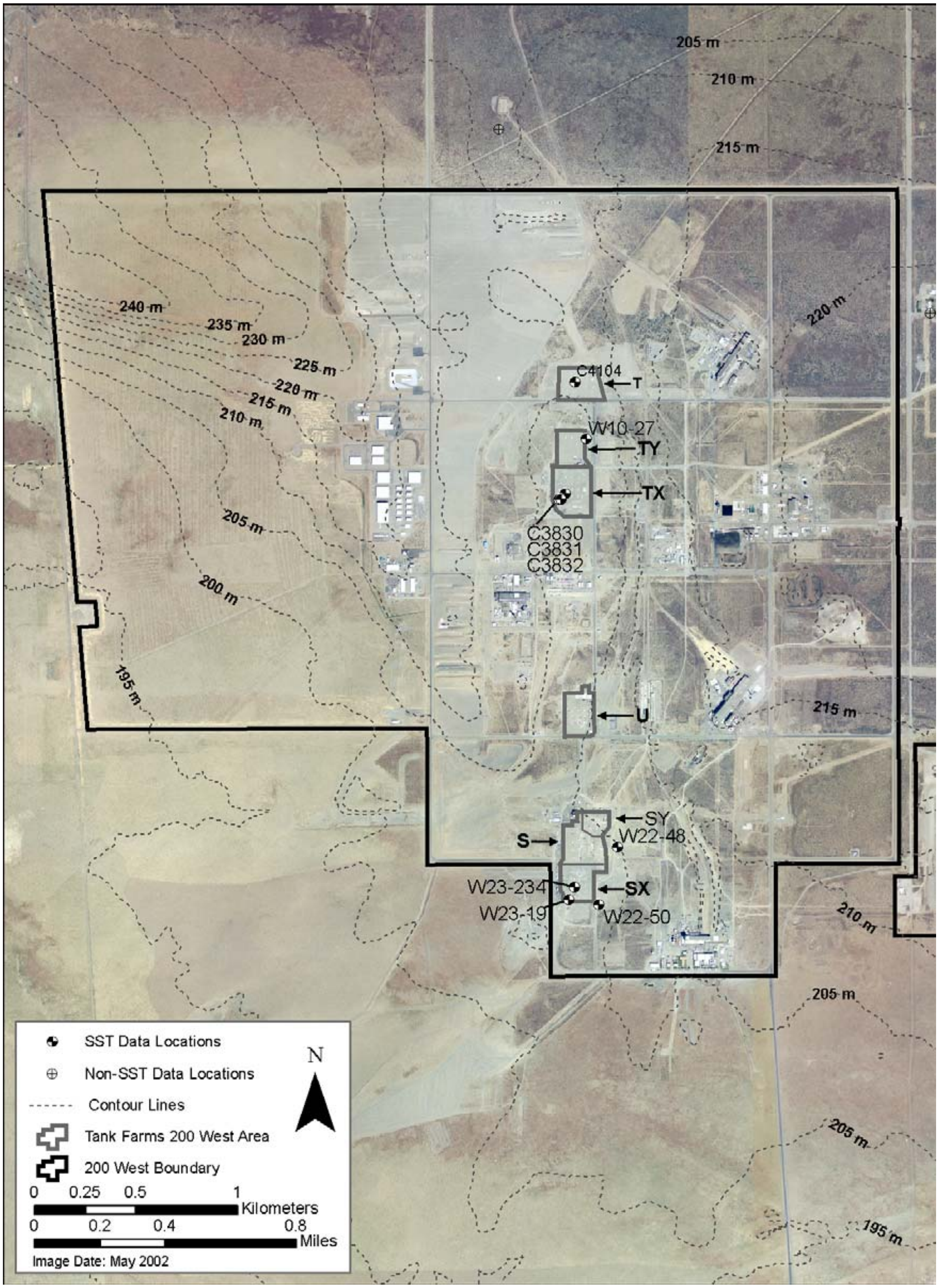

Figure 4.2. Topography of the 200 West Area

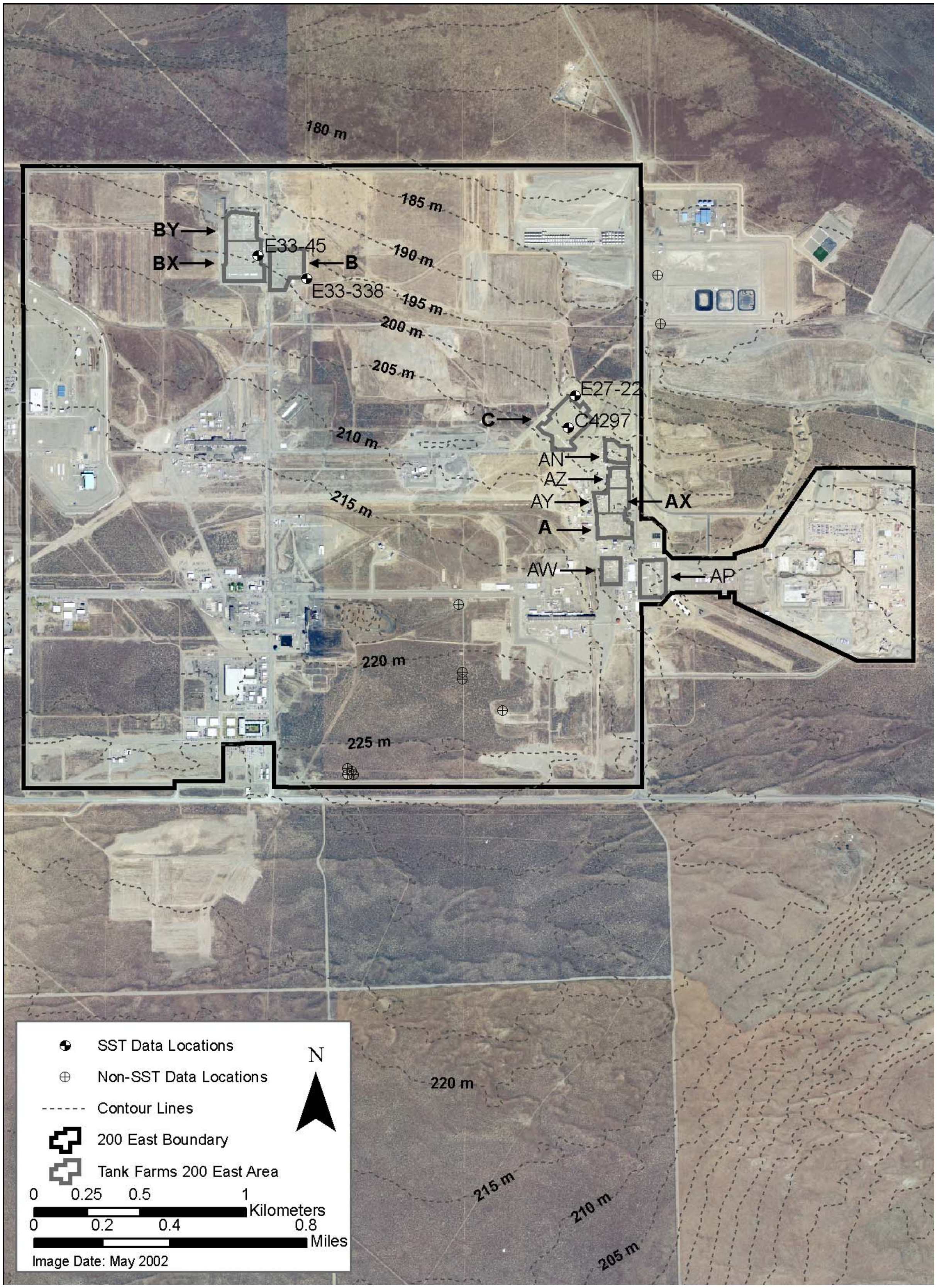

Figure 4.3. Topography of the 200 East Area

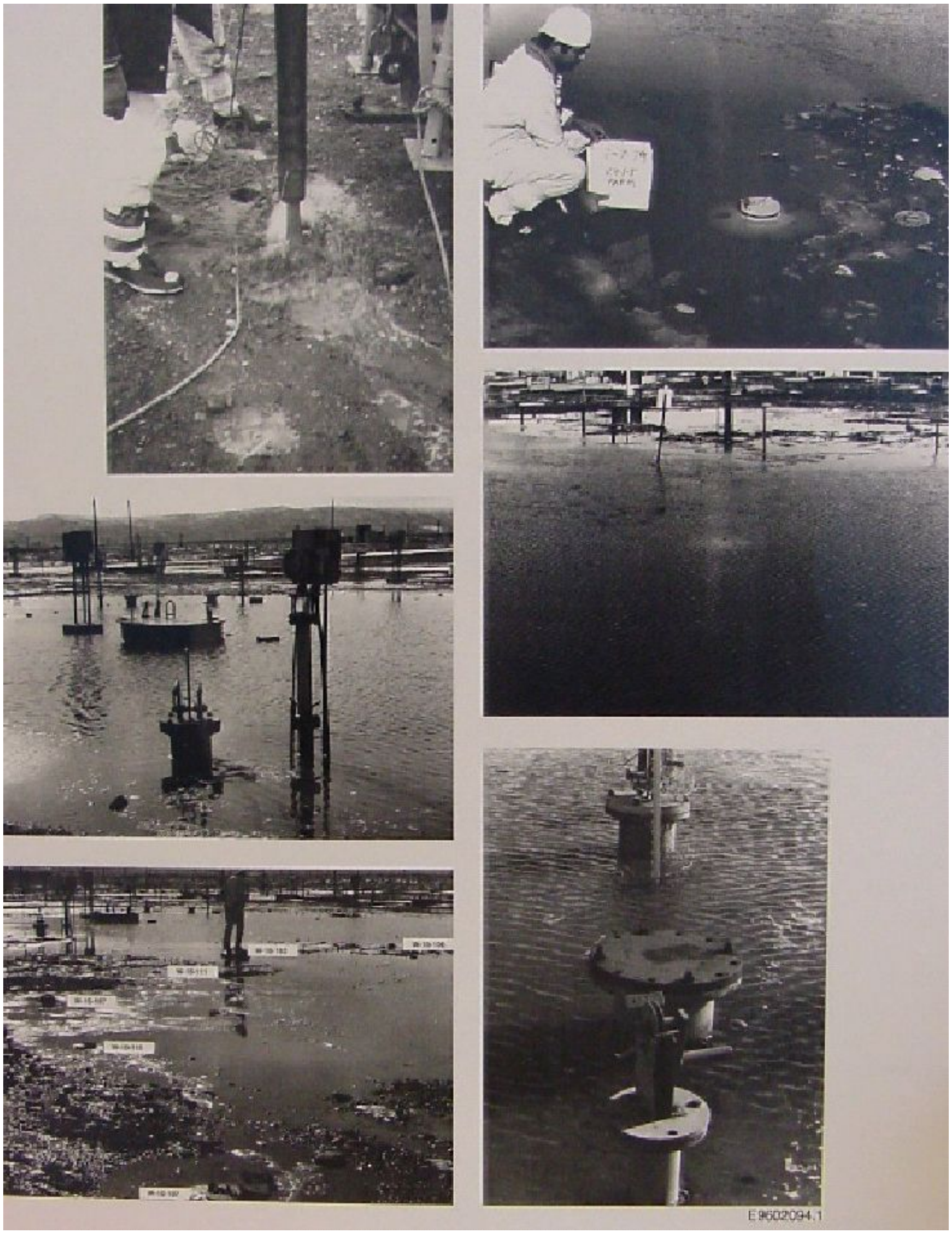

Figure 4.4. Temporary Flooding Above T Farm Following Rapid Snowmelt Above Frozen Ground (after Hodges 1998). Photograph is dated February 7, 1979. Hanford Meteorological Station weather records show air temperatures in January 1979 averaged −10.6°C (13°F), and on only one day did the maximum air temperature exceed 0°C (32°F).

4.2.4 Hydrology

As discussed in Section 3, runoff is rarely observed in natural systems at the Hanford Site and is usually not considered in recharge evaluations. However, in the highly developed areas of the Central

Plateau, the mix of paved, graveled, and compacted areas creates ideal conditions for initiating runoff and providing easy paths for overland flow. It took many years, but this runoff potential is now recognized and addressed (CH2M HILL 2002). Despite the best efforts, however, such events still occasionally occur and create problems. One recent example is the Spring 2004 thunderstorm that occurred at the BY Tank Farm in the 200 East Area. The intensity was very high. Although runoff was not observed in the surrounding terrain (given the absence of runoff features such as rills and gullies), the effect of runoff was readily apparent in and around the BY Tank Farm and the Prototype Hanford Barrier. Figures 4.5a and 4.5b show the upslope areas that contain bare soils that we suspect are highly compacted from vehicular traffic. We were surprised to see small erosion features on the tank farm surface, given our long-standing assumption that the surface is very gravelly. Once the water entered the narrow constriction between the barrier and the tank farm, velocity and depth likely increased, with the result that the erosive power was enough to gouge out the 1.0–m-deep gully seen in Figure 4.5c. Events such as these are highly transient and, we suspect, often not documented. The erosion caused by this particular event was not widely known until two weeks later.

4.2.5 Ecology

Flora and fauna are actively prevented from establishing in the SST farms to prevent the potential for mobilization of radionuclides to the environment. Activities to prevent establishment include periodic maintenance of the gravel cover and spraying for weed control. Even the areas surrounding the farms have marginal plant and animal activity because of the disturbed and/or compacted soils and vehicular traffic. Furthermore, those areas sometimes have gravel layers added to control dust and provide stable surfaces. Many of the SST farms are located within larger areas of disturbance that make it difficult for plant recruitment (when such is desired). Without sufficient plants for food and protection, establishment and survival of fauna is somewhat limited. Although one can find pockets of mature sagebrush here and there, the current landscape in the 200 Areas and especially in and around the tank farms is highly fragmented at best and not conducive to ecological health. One benefit of the large degree of disturbance is the ability to control the spread of fire more easily.

4.3 Current Status

Figures 4.6 through 4.11 show the visual condition of the SST farms as of May 2002. In all cases, it is evident that the farms themselves are devoid of vegetation and that significant areas surrounding the farms are either devoid of vegetation or covered with sparse plant communities. We anticipate that such conditions will continue as long as required to complete the clean out and closure of the tank farms. According to FIR documents (e.g., CH2M HILL 2002), emplacement of a final cover is not envisioned until the year 2040. That means the conditions seen in Figures 4.6 through 4.11 likely will remain for the next 33 years, except for those farms where an interim cover is placed to control infiltration of water (e.g., Zhang et al. 2007).

An unpublished data compilation[(a)] contains schematic maps of all SST farms showing the locations of all tanks as well as nearby burial grounds, cribs, trenches, ditches, french drains, retention basins, valve

(a) Chapter 3 – “Facility Description” (prepared by FM Mann and DA Myers). In *Single-Shell Tank RCRA Facility Investigation Report*, FM Mann (ed). CH2M HILL Hanford Group, Inc., Richland, Washington (title tentative; due to be published late 2007 or 2008.)

(a) (b)

(c)

Figure 4.5. Aftermath of Runoff Created by Spring 2004 Thunderstorm Near Waste Management Area BY. (a) View southeast of Prototype Hanford Barrier overlooking BY farm, source of some of the runoff; (b) view south of Prototype Hanford Barrier overlooking compacted area (other source of runoff) and western edge of BY farm (light-colored soil in foreground is where water ponded until it could pass through narrow constriction between barrier and tank farm; (c) view north showing erosion channel cut between barrier and BY farm (after Ward et al. 2005). Light-colored soil behind vehicle (c) marks where runoff water eventually ended up.

Figure 4.6. Waste Management Area A-AX. Also shown are double-shell tank farms AN, AP, AW, AY, and AZ. The tip of WMA C is just visible in the upper left corner of the image.

Figure 4.7. Waste Management Area B-BX-BY. The Prototype Hanford Barrier is the small rectangle located off the northwestern corner of the WMA BY.

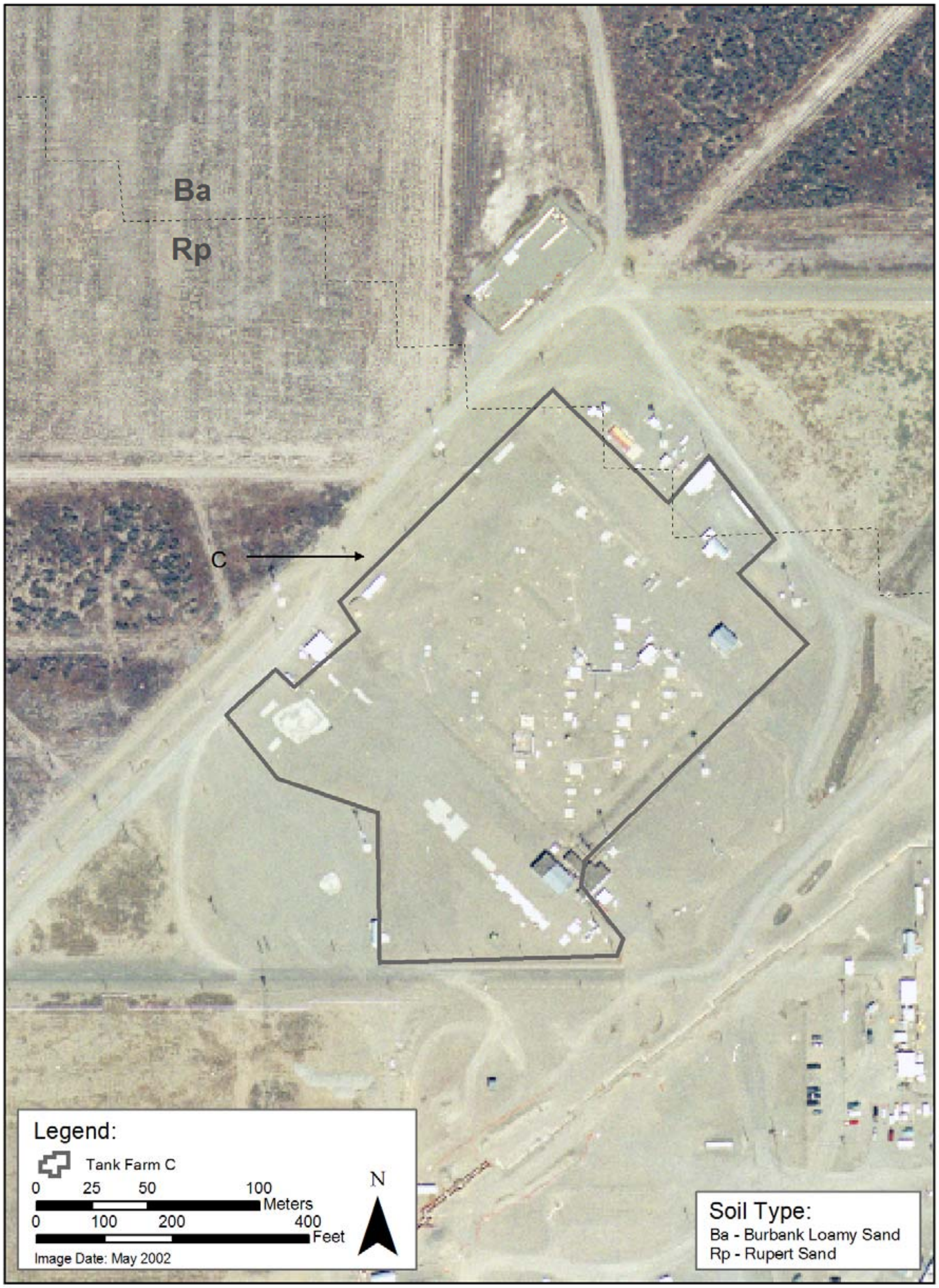

Figure 4.8. Waste Management Area C. Part of the double-shell AN Tank Farm can be seen in the lower right of the image. The 218-E-12A burial ground is the large rectangular area northwest of WMA C.

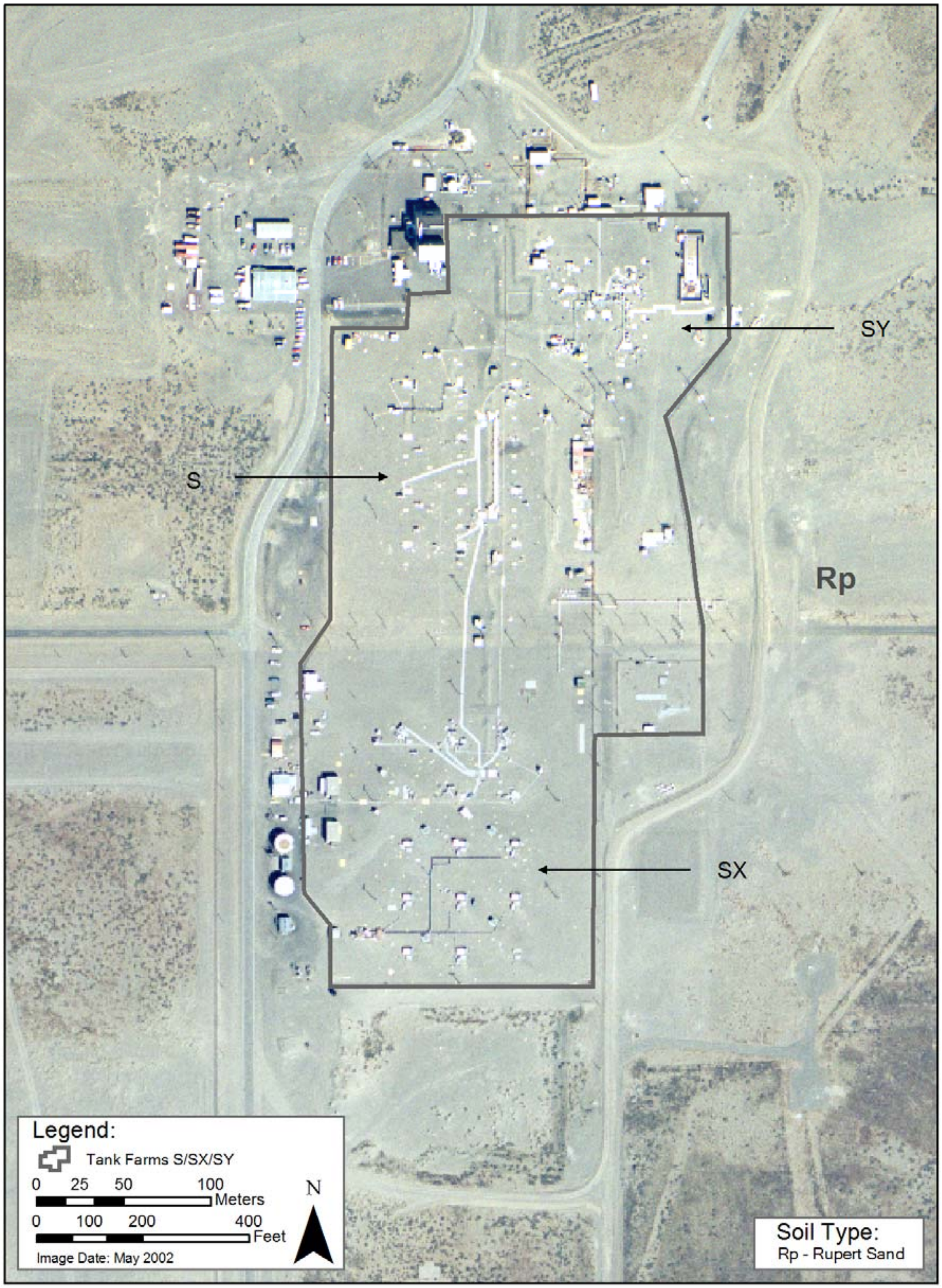

Figure 4.9. Waste Management Area S-SX

Figure 4.10. Waste Management Area T-TX-TY

Figure 4.11. Waste Management Area U

pits, vaults, tile fields, catch tanks, septic tanks, and other associated infrastructure. Most of these features are below ground and not visible in the images, and labels were not included so as to minimize clutter. We draw attention to their presence solely to alert the reader to the significant number of infrastructure elements below ground; the elements are not visible but may affect recharge and need to be considered when conducting tank-farm–specific assessments.

4.4 Additional Considerations

Several features of the SST WMAs could affect the analysis of recharge rates, including physical effects, temperature, and preferential flow. Physically, the tops of the SSTs are located between 1.8 and 2.7 m below the soil surface. A large impermeable surface at those depths has the potential to affect recharge by limiting the vertical movement of water, air, vapor, and heat. Under high recharge rates, water could collect on top of a tank and create a perched water table that, if near the soil surface, could allow for higher evaporation rates (and lower recharge rates) than would normally occur. There have been no reports of such ponding on the tanks, suggesting that water is not accumulating directly above the tanks. The sediments used as backfill are predominantly sand and gravel and are expected to be highly porous and conductive; thus, as water contents increase, the propensity for lateral flow increases. In all likelihood, water that collects on the tank top drains laterally until it reaches the edge of the tank and then it flows downward.

The impermeability of the tanks alters the movement of the air phase within the vadose zone. Because water vapor is always a component of the air phase, any alteration of air phase movement could influence the evaporation rate and thus affect recharge. For this report, we assumed that any alteration of air movement within the vadose zone is too small to affect recharge rates significantly.

Radionuclides within the SSTs undergo radioactive decay that generates heat and, in the past, has raised temperatures, often in excess of 100°C. The impact of this heat on recharge was examined for the S/SX FIR (CH2M HILL 2002), with the conclusion that increased temperatures would increase evaporation (and thus recharge) by no more than 2%. Appendix C provides more detail on the analysis used in the S/SX FIR. That analysis relied on several simplifications that may have affected the evaluation. The authors suggested that a more rigorous analysis could be employed in which the ground-surface temperature is computed using a radioactive-convective heat transfer approach, in which case the percentage of meteoric recharge lost through the ground surface would have been higher. Until such an evaluation is made, we have assumed that any temperature perturbation near the tanks has not affected recharge rates significantly. Outside the tank farms, the overall temperature gradient within the vadose zone is assumed to be too small to affect recharge rates significantly.

Preferential flow can occur at a very local scale (<10 cm) as a result of flow instabilities that lead to "fingering." Hendricks and Yao (1996) found that, for a sand dune in New Mexico, instabilities occurred during a precipitation event only when the total precipitation exceeded 4 cm. At Hanford, total precipitation in a 24-hour period has exceeded 4 cm only twice between 1947 and 2002 (by less than 1 cm in both cases). Therefore, flow instabilities were assumed to not be a dominant phenomenon affecting recharge at Hanford.

On a larger scale, actual preferential flow paths such as clastic dikes could affect recharge under the right conditions if they are present within 2 to 3 m of the soil surface. However, the soil and vadose zone sediment in and around the SST WMAs were significantly disturbed during construction such that any clastic dikes that may have been present were truncated at depth. Within the SST farms, the excavations were at least 12 m deep. Therefore, the assumption was made that clastic dikes would not be a factor in recharge directly within the SST farms. In the areas surrounding the SST farms and, more broadly, the 200 Areas, clastic dikes located within 3 to 4 m of the ground surface could affect recharge rates, but the degree of impact would be highly site-dependent and scenario-dependent (Murray et al. 2003; 2007).

Preferential flow could occur also as a result of focused overland flow. As mentioned in Section 3.5, the likelihood of overland flow in undisturbed areas is small except under conditions of rapid snowmelt above frozen soil, a condition that occurs infrequently at Hanford. Because of Hanford activities and infrastructure, overland flow is more common near and within WMAs and has been observed in the form of runoff from roads, parking lots, and compacted soil and flow of water off the roofs of structures. All of these processes are highly site-dependent and scenario-dependent.

5.0 Recharge Estimation Methods

Recharge rates at the Hanford Site can range from near zero to more than 100 mm/yr (Gee et al. 1992). To effectively cover this range, three complementary methods were used to estimate recharge rates in SST WMAs: lysimetry, tracers, and computer simulations. For a discussion of these and other methods, see the January–February 1994 issue of the *Soil Science Society of American Journal*, which contains a series of papers presented at a symposium titled "Recharge in Arid and Semiarid Regions." Scanlon (2004) provides an updated and broader review of methods for estimating both point and areal estimates of recharge. Rockhold et al. (1995) and Fayer and Szecsody (2004) described how some of these methods were used at Hanford. Much of those discussions are included here.

5.1 Lysimetry

The goal of lysimetry is to provide both performance data and model testing data for specific combinations of soil, vegetation, and precipitation. A lysimeter is a system that can be used to collect water that has infiltrated the soil and drained below the reach of plant roots and evaporation to become deep drainage and eventually recharge. Lysimetry is the only method available to directly measure deep drainage (which eventually becomes recharge). One of the strengths of lysimetry is that it can provide a control volume in which a number of water balance components can be measured directly. This control volume provides the data needed to calibrate numerical models that can then be used to forecast recharge.

The Hanford Site has used lysimeters for multiple purposes (Hsieh et al. 1973; Gee and Jones 1985; Freeman and Gee 1989; Wittreich and Wilson 1991; Gee et al. 1993; Ward and Gee 1997). The lysimeters used to provide data for this report include both containers that isolate the soil from its surroundings and field-scale pads that collect drainage but do not isolate the soil. Appendix A describes these facilities in detail. The major facilities are briefly described below.

A primary source of lysimeter data is the Field Lysimeter Test Facility (FLTF), which was constructed in FY 1987 to test the performance of capillary barrier designs (Gee et al. 1989; Fayer and Gee 2006). The FLTF contains 18 large lysimeters (surface areas of 2.3 and 3.1 m^2; depth from 1.5 to 3.0 m) and six smaller lysimeters (surface area is 0.07 m^2; depth 3.0 m). Treatments include variations of material types and thicknesses, the presence of vegetation, and the use of irrigation to mimic the increased precipitation of a possible future climate. Data from this facility include drainage, water content, matric potential, temperature, and vegetation observations.

Another source of lysimeter data for this report is the Prototype Hanford Barrier, a full-scale barrier constructed above an actual waste site (Wittreich et al. 2003). The Prototype Hanford Barrier design differs slightly from most of the tests in the FLTF in that the surface silt loam layer is 2 m thick (rather than 1.5 m) and the upper meter contains gravel for erosion control (only two FLTF lysimeters contained gravel in the silt loam layer, and only in the upper 0.3 m). More important, the Prototype Hanford Barrier differs from the FLTF tests in that it is a full-scale test that includes side slope effects.

The Prototype Hanford Barrier was instrumented to measure variables such as water content, matric potential, temperature, and drainage. One of the unique and valuable features of the Prototype Hanford Barrier is the presence of asphalt collection pads to collect drainage. These pads are part of the asphalt layer that underlies the entire barrier. Individual collection pads were constructed using asphalt curbing to separate the different collection zones. Four 322-m^2 collection zones underlie the main portion of the prototype barrier. Two similar zones lie beneath each of the two different side slope designs (one is sandy gravel, the other is basalt riprap). In addition, a collection lysimeter was constructed beneath the northeastern portion of the barrier, under the asphalt layer that lies beneath the basalt side slope treatment. This lysimeter provides a measure of the effectiveness of the asphalt layer in preventing drainage.

Although they provide the only direct measure of recharge, lysimeters have disadvantages. Lysimeters are usually fixed in space, which limits their ability to quantify the effects of spatial variability. The soil filling the lysimeter may not represent the natural stratification or layering that may be present. The length of record is much shorter than periods of interest. The lysimeter base and walls (if present) alter the natural gradients of temperature, airflow, and vapor flow that could be of importance when trying to measure recharge rates less than 1 mm/yr. When present, lysimeter walls restrict lateral root growth and promote downward growth. If plant roots reach the lysimeter base, the measurement of recharge can be compromised. When irrigated, lysimeters can be subject to the "oasis effect," in which heat from the unirrigated surroundings increases the evapotranspiration rate above what it would have been if the entire area had been irrigated. Finally, one of the issues with using lysimeters is verifying that no leaks of drainage water have occurred.

5.2 Tracers

The goal of the tracer method is to estimate historical recharge using measurements of tracer distributions in the soil and sediment of the vadose zone. The vertical distribution of tracers represents the integration of many recharge events and can be used to estimate the mean recharge rate for the time scale of interest for a performance assessment. Several tracers are available that enable estimates of recharge rates for durations of tens to thousands of years. The tracers examined for this report include chloride, chlorine-36, and the naturally occurring isotopes deuterium, oxygen-18, strontium, and uranium. Appendix B describes each method in detail. Each method is described briefly below.

Chloride originates from seawater spray that is carried by wind action into the atmosphere. The chloride is subsequently deposited naturally via precipitation and dry fall. Because of evaporation and plant water uptake, water below the evapotranspiration zone has an increased chloride concentration that is related to the deep-water flux, i.e., the recharge rate. With knowledge of the quantity of chloride introduced to the soil surface and information about the chloride profile in the subsurface, an estimation of recharge can be made:

$$J_R = \left(\frac{Cl_p}{Cl_s}\right) \times P \tag{5.1}$$

where J_R = recharge (mm/yr)
Cl_p = average chloride concentration in precipitation and dry fallout (mg/L)
Cl_s = average chloride concentration in soil pore water (mg/L)
P = average annual precipitation (mm/yr).

The product of Cl_p and P is termed the chloride deposition rate. The estimation of recharge using Equation (5.1) is commonly referred to as the chloride mass balance (CMB) method. Recharge estimates determined using Equation (5.1) may represent conditions spanning hundreds to thousands of years. Key assumptions are that chloride is nonreactive (i.e., no sorption or transformation) in the predominantly sandy sediments at Hanford, subsurface flow is plug-type flow, and precipitation and chloride deposition rates are steady during the time interval of interest.

Chlorine-36 originates from two sources: cosmic irradiation of atmospheric chloride and atmospheric testing of nuclear weapons (e.g., bomb-pulse chlorine-36). The quantities of chlorine-36 created by the testing were far higher than natural production rates and, thus, serve as a marker in the environment. The chlorine-36 data can be used to estimate the average recharge rate over the last 60 years.

Both chloride and chlorine-36 are conservative, nonvolatile, and almost completely retained in the soil when water evaporates or is transpired by plants (Phillips 1994). Some chloride is taken up by plants (e.g., Rickard and Vaughan 1988; Sheppard et al. 1998). Over hundreds to thousands of years, plant cycling is expected to have a minimal impact on the evolution of the chloride distribution in the soil profile beneath plants. Recharge rates determined with this method reflect conditions that existed hundreds to thousands of years ago and are sometimes called paleorecharge or paleofluxes. When such paleofluxes are used to represent current or future recharge conditions, the assumption is that the climate, soil, and vegetation conditions are similar. In contrast, bomb-pulse chlorine-36 has been around only ~60 years, so caution must be exercised when interpreting such data. In soils with high pH and high adsorption of other anions, anion exclusion can result in faster movement of chloride. Previous studies strongly suggest a relationship between soil surface area, which is determined primarily by clay content, and anion exclusion (e.g., Thomas and Swoboda 1970). Most of the sandy soils in and around the SST WMAs have relatively low percentages of clay, so the effects of anion exclusion in this soil should be relatively minor.

Two other issues that affect chloride-based estimates of recharge are mineral dissolution and the chloride dilution that is part of the measurement technique. Both issues can be significant when recharge rates exceed a few millimeters per year (Tyler et al. 1999; Gee et al. 2005).

Phillips (1994) suggested that systematic uncertainties in estimated chloride deposition rates can be as great as 20% if the chloride mass balance technique is extended to estimate recharge rates prior to the Holocene epoch (approximately 10,000 years ago). Scanlon (2000) suggested the uncertainty was as high as 38%. Because the Hanford Site was flooded by glacial meltwater about 13,000 years ago, the interpretation is not extended beyond that time. Therefore, the uncertainty in chloride deposition rates prior to Hanford Site operations is expected to be less than 38%.

There is some uncertainty about the local influence that Hanford Site operations may have had on the time-dependent concentrations of both chloride and chlorine-36 deposited at Hanford (Fayer et al. 1999). The concern about modern chloride deposition has not been resolved, but Murphy et al. (1991) examined the concern about chlorine-36 and concluded there was no nearby source that would confuse the chlorine-36 signal in the sediment.

Deuterium and oxygen-18 are inert isotopes of hydrogen and oxygen that are commonly used for recharge analysis because they are both stable and occur naturally in measurable quantities. As soil pore water evaporates, the isotopic composition of oxygen and hydrogen changes due to the preferential loss of lighter isotopes during evaporation. This results in the pore water having a larger ratio of heavier oxygen and hydrogen isotopes compared to the composition of precipitation. The magnitude of the isotopic shift is related to the level of pore water evaporative loss. The shift to heavier isotopes can be used to observe seasonal variations in water flux, identify the depth of evaporative enrichment, and estimate recharge.

The isotopic composition of oxygen and hydrogen are commonly reported relative to a standard material, using

$$\delta(‰)=\left(\frac{R_{sample}-R_{std}}{R_{std}}\right)\times 1000 \tag{5.2}$$

where R is the isotopic ratio (R = $^{18}O/^{16}O$ or $^{2}H/^{1}H$) and δ (delta or del) is the ratio of R of the sample to R of the standard material. Units of δ are permil (i.e., parts per thousand). Standard mean ocean water has been used as the standard material for past Hanford related oxygen-18 and deuterium recharge studies.

The oxygen-18 and deuterium isotopes are ideal for estimating recharge when evaporation is the only process removing water from the soil. In situations where plants are present, however, the method is more cumbersome. Plants extract water from the soil without discriminating between isotopes. To account for this extraction requires additional assumptions regarding plant water uptake behavior, and the additional assumptions increase the uncertainty of the recharge estimate. When plants are present but not considered in the analysis of recharge, the recharge estimates derived are considered upper boundary values.

Two relatively new environmental tracer methods for estimating recharge are to use ratios of strontium and uranium isotopes that occur naturally (Maher et al. 2003, 2006). From the ground surface to the ground water, the strontium isotopic ratio ($^{87}Sr/^{86}Sr$) of the pore water increasingly reflects the composition of the bulk sediment. The rate of change of the pore water strontium isotopic ratio towards the isotopic ratio of the bulk sediment, per specific depth interval, provides a measure of the pore water strontium flux relative to the sediment strontium dissolution flux. This rate of change can be used to estimate the ratio of the weathering rate to the fluid flux, from which the recharge rate can be estimated. Successful application of the method depends on the vadose zone being undisturbed by Hanford operations. As developed, the method provides estimates for rates that existed hundreds to thousands of years ago.

Like the strontium isotopic ratio, the uranium isotopic ratio ($^{234}U/^{238}U$) in sediment pore water is affected by the pore water flux and the dissolution rate of the bulk sediment. Unlike the strontium method, α-recoil loss of uranium-234 from the bulk sediment also contributes to the uranium isotope composition of the soil pore water. The α-recoil loss of uranium-234 is associated with the energetic α-decay of uranium-238. With information on the uranium isotope ratio profile, bulk weathering rate, and the rate of uranium-234 addition to the pore water by α-recoil, an estimate of recharge can be made. Successful application of the method depends on the vadose zone being undisturbed by Hanford operations. Both isotopic ratios (strontium and uranium) can be used to constrain (and thus improve) the recharge estimate (Maher et al. 2006).

5.3 Computer Simulation

The goals of simulation are to estimate recharge rates when there are little to no data and to leverage the existing short-term data into improved estimates of long-term recharge rates. Simulations of recharge at Hanford have been successful at highlighting the important factors that affect recharge and predicting recharge rates for specific cases. Simulation is the primary tool for forecasting recharge rates for future climate and land-use scenarios. The simulations also allow the results of the lysimetry and tracer methods to be merged on a consistent basis. Appendix C describes the simulation activity undertaken to estimate recharge rates for this report.

The UNSAT-H computer code was used to estimate recharge rates for this report (Fayer 2000). UNSAT-H can simulate nonisothermal water flow processes in both liquid and vapor phases and hysteresis in the soil hydraulic properties. This code has been tested using data from several of the lysimeter experiments at Hanford and elsewhere (Fayer et al. 1992; Fayer and Gee 1992; Khire et al. 1997; Andraski and Jacobson 2000; Scanlon et al. 2002). Fayer and Gee (1997) tested UNSAT-H and another code and concluded that UNSAT-H provided far better estimates of drainage through a surface barrier. Scanlon (1992) used a similar unsaturated flow code to estimate recharge rates in the Chihuahuan Desert of Texas. Scanlon et al. (2002) tested the code with lysimeter data from two sites.

One of the disadvantages of numerical simulation is that it requires numerous parameters to represent climate, soil, and vegetation characteristics. In many instances, these parameters are unknown or known only marginally. Another disadvantage is the use of conceptual simplifications to make the simulation tractable. Numerical simulation with a code such as UNSAT-H is the most flexible method for estimating recharge rates, but its data-intensive needs and conceptual simplification could lead to recharge estimates that have the most uncertainty.

6.0 Results

This section estimates recharge for soil and vegetation conditions specific to the WMAs. In cases where multiple estimates are available, a best (or mean) rate is assigned. The recharge estimates are based on the latest published information and can take the place of estimates provided by Last et al. (2006) for comparable soil-vegetation combinations. The soil types were evaluated for conditions that existed during four periods—prior to Hanford, during disposal facility operations, during the design life of a surface barrier, and following the design life of a surface barrier.

6.1 Integration of Recharge Estimates for Each Soil Type

Recharge rate estimates were evaluated for five natural soil types, two modified soils, a nominal barrier design, and a nominal barrier side slope design:

- Rupert sand
- Burbank loamy sand
- Ephrata sandy loam
- Hezel sand
- Esquatzel silt loam
- Hanford formation sediment
- graveled surface
- surface barrier
- side slope of surface barrier.

Presented below are the estimates that represent what is reasonably expected based on the available data; they are not estimates of the lowest or the most conservative recharge rates. For each soil type, the data available for estimating recharge rates were assessed. Where data are conflicting, alternative recharge estimates are presented. This report does not address the variability in recharge estimates, did not conduct sensitivity tests, and did not consider alternative conceptual models, as done by Fayer and Szecsody (2004) for the IDF. All are valid and relevant topics but outside the scope of this report.

6.1.1 Rupert Sand

Rupert sand is the most prevalent soil type in both the 200 Areas as well as to the south of the 200 Areas. Several estimates of recharge in vegetated Rupert sand are available, Appendix A describes the work of Enfield et al. (1973), who estimated a recharge rate of 0.26 mm/yr at the 200 East deep well site, which is located about 2 km south of the IDF. They derived the estimate from measurements of temperature and matric potential gradients and estimates of the soil hydraulic properties. Enfield et al. (1973) cautioned that the estimate was tentative, given the uncertainty in the hydraulic properties.

Appendix B describes several tracer estimates for Rupert sand with plants. Murphy et al. (1996) estimated a rate of 4.0 mm/yr for a borehole near the Wye Barricade (about 11 km southeast of the 200 East Area). Fayer and Szecsody (2004) estimated a rate of 0.9 mm/yr for soils at the IDF site (southcentral portion of the 200 East Area). The value for the IDF site was derived by averaging recharge estimates determined using chloride data from seven boreholes. Fayer and Szecsody (2004) used deeper chloride data because of concern that some of the shallower chloride originated from coal-plant emissions and would have biased the estimates toward much lower values of recharge. Vegetation at the IDF was described as being healthy mature shrub-steppe. Vegetation at the Wye Barricade site was described as predominantly cheatgrass and Sandberg's bluegrass and a sparse cover of gray rabbitbrush and sagebrush. Murphy et al. (1996) noted that the sparse cover of shrubs may have been the result of past range fires.

Appendix C describes the simulation results for Rupert sand. For the case with a shrub cover, the simulation yielded a rate of 1.8 mm/yr. When plants were absent, the simulation yielded a value of 45 mm/yr, which is nearly identical to the value of 44 mm/yr recommended by Last et al. (2006). The major difference is that the simulation in Appendix C used the entire 50-year HMS weather record, which includes the two wettest years (1995, 1996) on record.

To provide an estimate of recharge, we treated the IDF recharge estimate as a single value (0.9) to avoid weighting that site too heavily and averaged it with the Wye Barricade estimate (4.0) and the estimate from the 200 East deep well test (0.26). The result, which we recommend using, is a recharge estimate for vegetated Rupert sand of 1.7 mm/yr. For the case of Rupert sand without vegetation, we recommend using the rate of 45 mm/yr.

6.1.2 Burbank Loamy Sand

Next to Rupert sand, Burbank loamy sand is the second most prevalent soil type in both 200 Areas as well as to the north of the 200 Areas. There are no direct measurements of recharge rates in vegetated Burbank loamy sand, but there are tracer-based estimates. As Appendix B notes, there is some uncertainty about the soil type at specific borehole locations, so for this data package, all boreholes that might be in Burbank loamy sand were included. A recharge estimate of 1.9 mm/yr resulted from averaging the five Prych (1998) estimates (2.8, 5.5, 1.8, 1.2, and 0.66 mm/yr), the two Fayer and Szecsody (2004) estimates (0.16 and 0.24 mm/yr), and the Keller et al. (2007) estimate (2.5 mm/yr). An alternative averaging scheme is to group nearby boreholes into a single estimate to lessen the impact of any specific area. In this scheme, Prych (1998) boreholes B10 and B12 were averaged, Prych boreholes B17 and B18 were averaged, and the two Fayer and Szecsody (2004) boreholes were averaged. The result was a recharge estimate of 1.8 mm/yr, which is nearly identical to the simpler averaging scheme. The simulation results in Appendix C suggest a rate of 5.0 mm/yr, which is near the upper end of the range of tracer estimates. Given that the tracer estimates are likely to give a better estimate of recharge, all else being equal, our recommendation is to use the simple average tracer-based estimate of recharge, i.e., 1.9 mm/yr for vegetated conditions

The simulation results in Appendix C for the unvegetated Burbank loamy sand yielded a rate of 53 mm/yr. This value is only slightly higher than the value of 52 mm/yr recommended by Last et al. (2006). The major difference is that the simulation in Appendix C used the entire 50-year HMS weather record, which includes the two wettest years (1995, 1996) on record. We recommend using the rate of 53 mm/yr.

6.1.3 Ephrata Sandy Loam

Ephrata sandy loam occupies the northern part of the 200 East Area and is not present in the 200 West Area. This soil type is more common north of the 200 Areas. The WMA B-BX-BY is the only WMA located within the boundaries of this soil type. Last et al. (2006) recommended a rate of 1.5 mm/yr when there was shrub vegetation, based on the average of two tracer-based estimates by Prych (1998). As

Appendix B notes, there is some uncertainty about the soil type at specific borehole locations, so for this data package, all boreholes that might be in Ephrata sandy loam were included. A recharge estimate of 2.8 mm/yr for vegetated Ephrata sandy loam resulted from averaging the four Prych (1998) estimates (2.8, 5.5, 1.8, and 1.2 mm/yr) and the Keller et al. (2006) estimate (2.5 mm/yr). The estimate of 2.8 mm/yr is nearly double that of Last et al. (2006), our estimate includes more boreholes and so should be considered to be more representative. Appendix C shows that the simulation estimate of recharge was 0.2 mm/yr, which is close to the low end of the tracer estimates. Given that the tracer estimates are likely to give a better estimate of recharge, all else being equal, our recommendation is to use the simple average tracer-based estimate of recharge, i.e., 2.8 mm/yr. If it is desired to treat the Burbank and Ephrata soils as being identical and specifying a single rate, that rate should be 1.9 mm/yr because it was derived by averaging the rates from all boreholes.

Last et al. (2006) recommended a rate of 17 mm/yr when there was no vegetation, based on simulation results in which weather data from 1958 to 1992 were used. Appendix C extends those simulations to 2006, with the result that recharge was estimated to be 23 mm/yr when plants are absent. We recommend the rate of 23 mm/yr be used when plants are not present.

6.1.4 Hezel Sand

Hezel sand occupies a small portion of the southwest corner of the 200 West Area. The nearest SST farm is SX, which is located about 1.5 km to the east. There are no tracer or measurement data for this soil type. The simulation results in Appendix C showed a rate of <0.1 mm/yr with shrubs and 8.7 mm/yr without plants. Because Last et al. (2006) do not provide estimates for this soil type, we recommend using our estimates for Hezel sand.

6.1.5 Esquatzel Silt Loam

Esquatzel silt loam occupies a small portion of the southwestern corner of the 200 West Area. The nearest SST farm is U, which is about 1.5 km to the east. There are no tracer or measurement data for this soil type. The simulation results in Appendix C showed a rate of <0.1 mm/yr with shrubs and 8.6 mm/yr without plants. The rate in the absence of plants seems high for a silt loam soil, but no other estimates for this condition exist. The low rate predicted with shrubs is consistent with recharge estimates for other silt loam soils at Hanford (e.g., Murphy et al. 1996; Prych 1998). Because Last et al. (2006) do not provide estimates for this soil type, we recommend using our estimates for Esquatzel silt loam.

6.1.6 Hanford Formation Sediment

Throughout the lifetime of the Hanford Site, construction and operation activities have repeatedly scraped off the thin veneer of real soil and exposed Hanford formation sands. These sediments tend to be coarser than the original soil, and plants have difficulty reestablishing on them. Over time, surface soils across much of the 200 East and West Areas have been transformed into something more accurately described as Hanford formation sands (generically called backfill) rather than the original designations (e.g., Rupert sand; Burbank loamy sand). In addition, the excavation process broke up any near-surface layering and clastic dikes that may have potentially affected recharge rates (see Section 3.2 for more detail). In these situations, the outcome is a new soil type that has a relatively uniform composition and lacks any clearly distinguishable layers that might impede downward movement of water.

There are no data for recharge in Hanford formation sediment with plants. This is understandable, given that most disturbed areas are in or near controlled radiological zones and plants tend to be excluded to prevent contaminant uptake. The simulation results in Appendix C provide two recharge estimates for the case with shrubs: 1.8 and 31 mm/yr. The difference between the two reflects the different hydraulic properties assumed for the soil. Given the highly uncertain nature of the hydraulic properties, we are not providing a recommendation for recharge for this situation.

Appendix A describes the data collected from the 300 North Lysimeter in the 300 Area. For a nearly 22-year period, drainage from the 300 North Lysimeter averaged 62 mm/yr (Gee et al. 2005a). Some of that period included the two wettest years (1995, 1996) on record, so the average may be biased high relative to a long-term precipitation rate. Regardless, the record shows that coarse sand without vegetation can result in a relatively high recharge rate. Appendix C describes the simulation results for this soil type without plants. The predicted rates were 22 and 91 mm/yr for the two soils evaluated. These estimates bracket the measured value of 62 mm/yr. This value is nearly identical to the value of 63 mm/yr recommended by Last et al. (2006). We recommend using the value of 62 mm/yr to represent recharge for unvegetated Hanford formation sand.

The nature and properties of Hanford sand will change as a result of the inexorable soil development that will occur once it has been left alone and undisturbed. Hanford sand is a new soil type. As it matures, this soil type should see increased levels of organic matter, increased levels of fine-textured materials such as silt and clay, and a deepening of the soil profile. All of these changes suggest that recharge rates under Hanford sand will slowly decrease.

6.1.7 Graveled Surface

Fayer and Szecsody (2004) is the most recent publication with data appropriate to this soil type when plants are not present. Last et al. (2006) used those data to estimate a recharge rate of 92 mm/yr. Their estimate was derived using measured drainage from two lysimeters and precipitation received during the measurement period and scaling the measured drainage linearly to the long-term average precipitation of 173 mm/yr. Data collected since that publication have not been analyzed or published.

One of the potential serendipitous benefits of the SST borehole data packages was that the reported anion data could be used to estimate recharge rates for graveled surfaces. Appendix B explains in detail how the data packages were reviewed and analyzed. For several reasons, including sampling and measurement issues, we concluded that the data were sufficiently inconclusive and should not be used for recharge estimates for the SST RFI.

Two tracer studies were conducted using techniques not used previously at Hanford on samples from a borehole just to the east of the WMA S-SX. DePaolo et al. (2004) used oxygen-18 to infer recharge rates of 35 to 60 mm/yr. These are consistent with our expectation for graveled surfaces, but we felt that the newness of the application to Hanford required more consideration before acceptance. Maher et al. (2006) used a methodology that employed natural isotopic ratios of strontium and uranium and estimated a rate of 5 ± 2 mm/yr. This estimate is applicable to recharge hundreds to thousands of years ago in undisturbed sediments and is consistent with current estimates using chloride under these conditions. However, it does not provide an estimate of the recharge rate that resulted from the presence of the graveled surface.

In lieu of additional data, we recommend that the 92-mm/yr rate suggested by Last et al. (2006) be used.

6.1.8 Surface Barrier

Surface barriers are envisioned for many of the waste sites in the 200 East and West Areas. However, which designs will be used for which waste sites are yet to be determined. For this report, we assumed most barriers will be designed to be functionally equivalent to the Hanford Barrier. Fayer and Gee (2006) have shown that this design, which uses silt loam soil for the surface layer, limits recharge to less than 0.1 mm/yr under a variety of scenarios, including the complete lack of vegetation. Field measurements, tracer studies, and computer simulation support these low rates (e.g., Fayer and Szecsody 2004; Ward et al. 2005; Fayer and Gee 2006). We recommend continuing to use this estimate until a formal barrier design process identifies actual designs that can be evaluated.

6.1.9 Side Slope of Surface Barrier

Last et al. (2006) recommended using a recharge estimate of 3 mm/yr for the barrier side slope with a fully developed shrub community. That recommendation was based on the assumption that a sandy gravel/gravelly sand side slope would eventually develop to resemble a Burbank loamy sand, for which Last et al. (2006) have estimated a recharge rate of 3 mm/yr (based on chloride data from three of the five boreholes discussed by Prych [1998]). Assuming that side slopes do evolve to resemble Burbank loamy sand, we recommend using the value proposed for that soil type, namely 1.9 mm/yr. As we learn more about the final design of side slopes and how they age and evolve, this estimate can be updated.

Last et al. (2006) provided a recharge estimate of 33 mm/yr for barrier side slope that is mostly unvegetated. That estimate was based on 10 years of measurements (November 1994–September 2004) on the side slope of the Prototype Hanford Barrier (Ward et al. 2005). Because precipitation received during that period exceeded the long-term average, the measured drainage was scaled to the long-term precipitation rate of 173 mm/yr. Data collected after publication of Ward et al. (2005) have not been analyzed or published.

In contrast to the drainage data from the prototype side slope, data from lysimeter D4 at the FLTF yielded a drainage rate of 113 mm/yr. When corrected for precipitation received during the monitoring period, the long-term rate was estimated to be 100 mm/yr (Fayer and Szecsody 2004). Both values are much higher than drainage from the prototype, even though the source of sediment in the lysimeter was the stockpile used for the prototype side slope. The inconsistency may be the result of unique features of the prototype (see next paragraph), but it could also reflect variability in the sediment stockpile such that D4 is not representative of the bulk of the side slope.

Also in contrast to the drainage rate for the prototype side slope is the drainage rate of 62 mm/yr from the 300 N lysimeter. As described in Appendix A of Fayer and Keller (2007), the 300 N lysimeter is filled with Hanford formation sand from which gravel particles were mostly screened out. All else being equal, we would expect sand to have a lower drainage rate than sandy gravel. In this case, the opposite occurred. Again, the inconsistency may be the result of unique features of the prototype (see next paragraph), but it may also reflect differences in the physical and hydraulic properties of the two tests.

Appendix A discusses some of the challenges with using the prototype data set to represent an unvegetated side slope, challenges that include the increasing vegetation cover, the record-setting precipitation rates during the early years when there was no vegetation, the presence of a graveled road over the lysimeter, and the lack of controls on root access to the lysimeter basin. Despite these concerns, this data set is the best in hand because it comes from a real side slope that was constructed will full-scale earth-moving equipment. Therefore, we recommend continuing to use the estimate of 33 mm/yr for the unvegetated side slope of a surface barrier. At the same time, we also recommend that the inconsistencies between the lysimeters and prototype be resolved.

6.2 Summary of Recharge Estimates

Table 6.1 summarizes the estimated recharge rates recommended for each soil type and vegetation condition. Those recommendations that differ from Last et al. (2006) are listed in bold type.

For the period prior to Hanford, the natural soils were assumed to be undisturbed and similar to what exists currently. The vegetation was assumed to be a healthy shrub-steppe with a mixture of grasses and shrubs. During the past few thousand years, the Central Plateau was assumed to have experienced normal weather cycles, vegetation changes in response to fire, drought, disease, and pests, and soil development. The water and geochemical conditions observed in the present-day vadose zone were assumed to be a consequence of these assumptions about past soil and plant conditions.

For the period during disposal operations, wherever Hanford formation sediments were exposed, recharge rates for this soil should be used. Wherever the natural soil remains undisturbed, we recommend using same rate as the pre-Hanford state. The same recommendation holds for the period during design life of any barrier.

For the period after the design life of any surface barrier, external changes in climate and vegetation were assumed to continue to influence the surface barriers and surrounding soils but not measurably change the recharge rates in any negative way (i.e., increase recharge). The surrounding soils were assumed to behave in a fashion similar to their pre-Hanford state.

The recommendations provided in this report are based on published data, which is limited, given the variety of soil and vegetation conditions and the myriad scenarios that might be considered. Necessarily, a sensitivity analysis was not performed and alternative conceptual models were not evaluated. However, the reader can get a sense of sensitivity and uncertainty related to recharge by reviewing the work done for the IDF site (Fayer and Szecsody 2004). They considered vegetation presence, type, and abundance; soil properties; climate; dune sand migration; and anticipated farming practices.

6.3 Recommendations

During preparation of the recharge data package for the IDF, Fayer and Szecsody (2004) noted a number of issues that could affect their estimates of recharge rates. They included climate change, bioturbation (plant and animal mixing of soil), unstable and preferential flow, flaws in a barrier (e.g., differential settling and cracking; discontinuities; points of flow convergence), possible facility deposition of chloride, and the importance of temperature and water vapor flow when recharge rates are

Table 6.1. Estimated Long-Term Drainage Rates for Use in Hanford Assessments (supplements rates provided by Last et al. 2006). Items in bold text indicate revisions to Table 2.1 of Last et al. (2006); recharge rates for Hezel sand and Esquatzel silt loam are new.

Soil Type	Estimated Long-Term Drainage Rates (mm/yr)	
	Shrub	No Plants
Rupert sand (near U.S. Ecology)	5.0	30
Rupert sand (near IDF)	0.9	**45**
Rupert sand (elsewhere on Central Plateau)	**1.7**	**45**
Burbank loamy sand	**1.9**	**53**
Ephrata sandy loam	**2.8**	**23**
Hezel sand	**<0.1**	**8.7**
Esquatzel silt loam	**<0.1**	**8.6**
Hanford formation sand	Np	**62**
Graveled surface	Np	92
Modified RCRA C barrier	0.1	0.1
Gravel side slope on surface barrier	**1.9**	33[(a)]

np = Not provided by Last et al. (2006) or this report.
(a) Tentative because of concerns regarding presence of some plants, road, and small section of vegetated silt-loam within the drainage collection zone.

lower than 1 mm/yr. They highlighted the need to estimate the longevity of any surface barrier. They recommended analogue studies to strengthen the understanding of long-term changes and help to reduce the uncertainty embodied in the predictions.

In the same vein, preparation of this report caused us to note a number of additional issues that are just as important. They include the lack of any physical and hydraulic characterization data for SST WMAs, the nature of the WMAs in the next 30 years, and the speed and character of restoration. Where possible, we provide recommendations for addressing them (see below).

A key issue is the lack of any physical and hydraulic characterization data for the near-surface material in SST WMAs. These data are needed to predict recharge rates for each SST farm because it is visually clear that the surface material differs from SST farm to SST farm. The data are needed also to be able assign recharge rates based on lysimeter data for specific materials. Sampling within tank farms is recognizably problematic, but it can be done (just as borehole samples were collected from deeper depths for geochemistry studies).

Given the current state of the SST WMAs and the likelihood that they will look nearly the same for the next 30 years, we believe the rates for unvegetated Hanford sand will continue to be important. However, we believe that *Hanford sand* is too broad a term to use and that a better method of differentiating the material on a site-specific basis is warranted. Furthermore, the impact of gravel additions for stabilization purposes must be considered (Parsons 1996).

Finally, little is known about the speed and character of restoration, particularly with respect to soils that have been extremely disturbed. We need to understand and be able to predict how disturbed and graveled soils alter recharge rates, how easily they allow vegetation to reestablish, and how soon they begin to develop true soil properties such that they start to resemble natural soils. Shafer et al. (2007)

considered these questions at the Nevada Test Site and described a methodology that could be used at Hanford. Related to soil development is the potential for other processes (e.g., sand dune migration into WMAs) to occur that could affect conditions in the WMAs. If such processes are sufficiently likely, we need to be able to predict how the changes caused by those processes will impact recharge rates.

7.0 Conclusions

The objective of this data package was to use published data to provide recharge estimates for the scenarios being considered in the RCRA Facility Investigation for SST WMAs being prepared by CH2M HILL. Recharge rates were estimated for areas that remain natural and undisturbed, areas where the vegetation has been disturbed, areas where both the vegetation and the soil have been disturbed, and areas that are engineered (e.g., surface barrier). The recharge estimates supplement the estimates provided by Last et al. (2006) for the Hanford Site using additional field measurements and computer simulations using weather data through 2006.

One key conclusion important for evaluation of recharge in SST WMAs is that the data in the borehole characterization reports to date are not sufficient for recharge estimation. Recharge evaluation requires specific data types, spatial resolution, measurement techniques, and system knowledge. Because recharge estimation was not identified as a critical need during the data quality objective (DQO) process, these requirements for recharge estimation were not considered when the SST borehole characterization efforts were planned and executed.

Several issues were identified for resolution. They include the lack of physical and hydraulic characterization data for the near-surface material in tank farms, the lack of characterization information for Hanford sand on a site-specific basis, and the gap in knowledge about the speed and character of soil and plant community restoration, particularly with respect to soils that have been extremely disturbed. The gap in knowledge also extends to understanding the potential for other processes (e.g., sand dune migration into WMAs) to occur that could affect recharge conditions in the WMAs, including the performance of surface barriers.

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Appendix A

Measurements for Recharge Estimation

Appendix A

Measurements for Recharge Estimation

The strongest estimates of Hanford-era recharge rates are those based on actual water balance measurements under conditions at the Hanford Site. They include direct measurements of drainage from lysimeters or measurements of other processes that can be used to estimate recharge. For some combinations of soil and plants, measurements may be required for decades or longer because the drainage rate is so low. In such cases, lysimeters may need to be monitored for decades before low rates of recharge can be measured reliably. In contrast, in disturbed areas where recharge is expected to be higher, lysimeter measurement techniques can be ideal for obtaining accurate estimates of current recharge rates under well-defined conditions.

Section 5.1 discusses the advantages and disadvantages of lysimeter measurements. One issue of lysimeter design that is critical to recharge estimation is the depth of the lower boundary relative to the evapotranspiration zone. The lower boundary needs to be sufficiently deep that evapotranspiration is unaffected by its presence; otherwise, drainage measurements will be affected. For most of the facilities described in this Appendix, the lower boundary is sufficiently deep. For those few cases where the boundary may not be deep enough, an explanation is provided.

This appendix summarizes the available water-balance data for estimating recharge at the Hanford Site. The appendix is organized in sections, one section for each measurement site. Where appropriate, the descriptions from the original texts were used. Figure A.1 shows the location of each site relative to the Central Plateau.

A.1 Field Lysimeter Test Facility

The Field Lysimeter Test Facility (FLTF) was constructed as part of the Protective Barrier Program in FY 1987 to test the performance of capillary barrier cover designs (Gee et al. 1989a; Wing 1994). Figure A.1 shows the location of the FLTF within the 200 Areas. After 1994, the facility began being used to conduct testing of soil-vegetation-climate treatments of importance to performance assessments of the Integrated Disposal Facility (IDF).

Figure A.2 shows that the FLTF contains 24 lysimeters of three types—14 drainage, 4 weighing, and 6 small-tube lysimeters. The drainage lysimeters are vertical cylinders that are 3 m deep and 2 m in diameter (surface area of 3.1 m^2). The drainage lysimeters compose the walls of the FLTF. The weighing lysimeters are boxes with length and width dimensions of 1.5 m and a depth of 1.7 m (surface area of 2.3 m^2). The boxes rest on platform scales to enable hourly weight measurements of water gain and loss. The small-tube lysimeters are vertical cylinders that are 3 m deep and 0.3 m in diameter (surface area of 0.07 m^2). Unlike the others, the small-tube lysimeters are clear Plexiglas to facilitate root and soil observations. These lysimeters are arrayed along the inner walls of the FLTF.

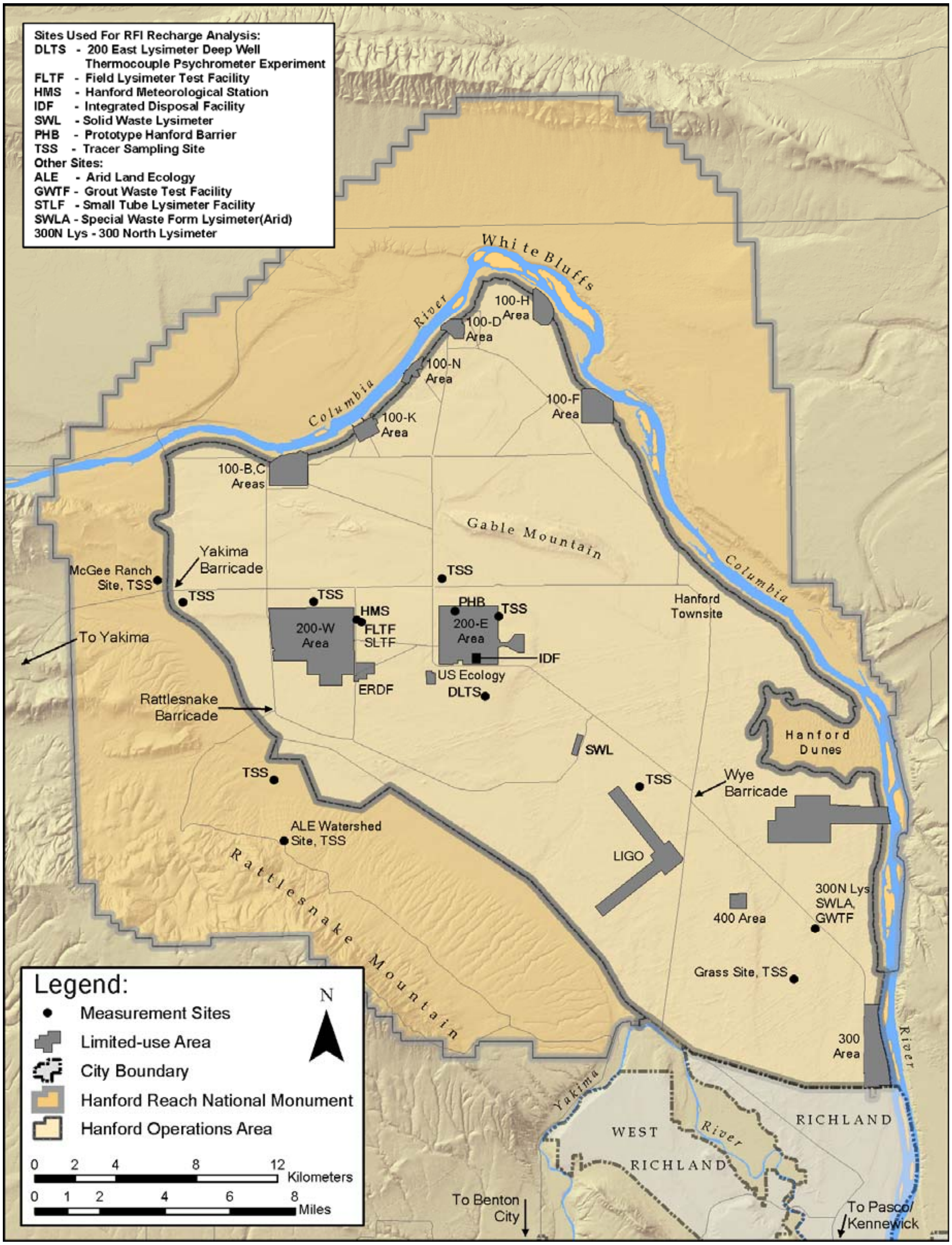

Figure A.1. Locations of the Measurement Sites

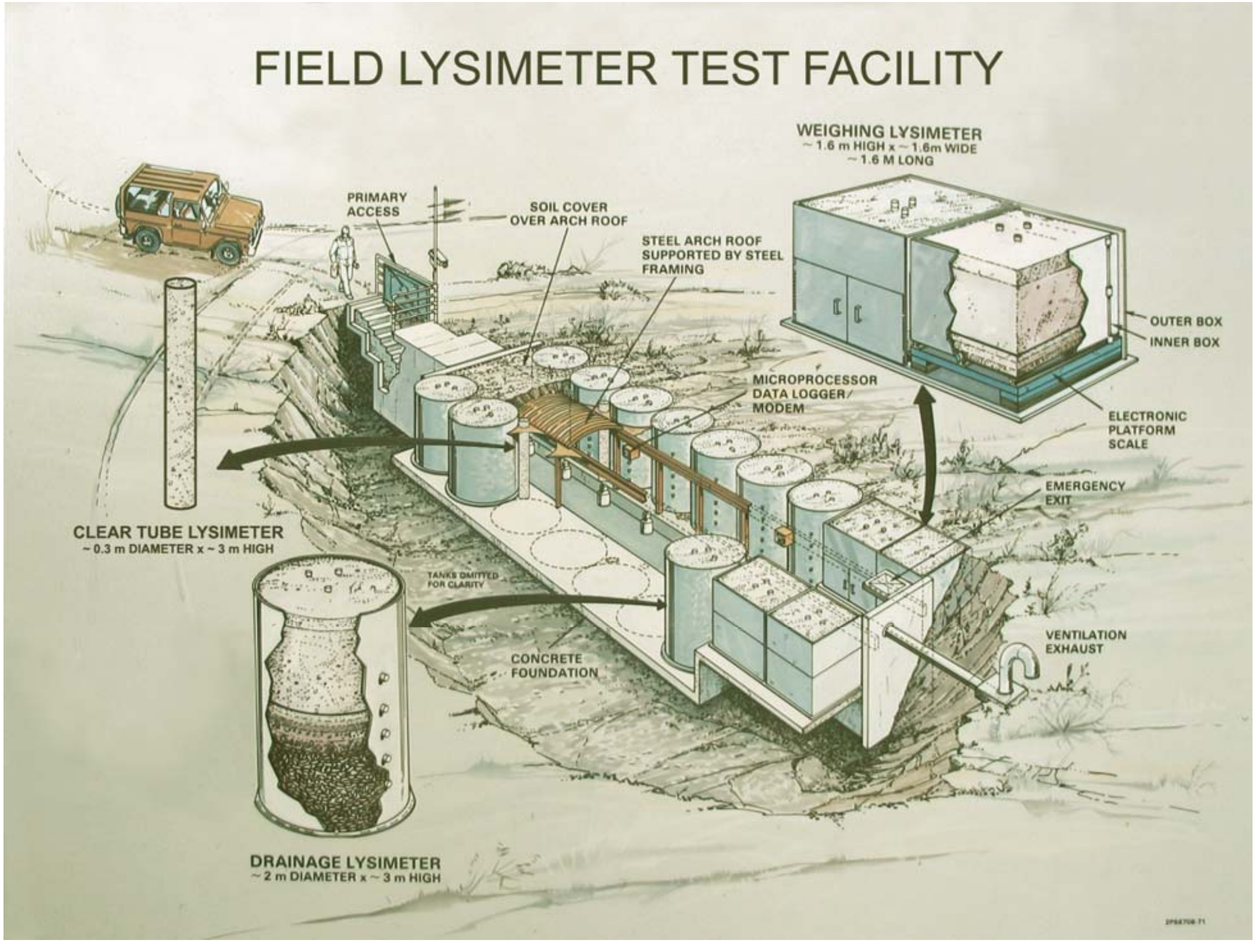

Figure A.2. Artist's Rendering of the Field Lysimeter Test Facility

Treatments include variations of material types and thickness, presence of vegetation, and the use of irrigation to mimic the possible increased precipitation of future climate. The data collected from this facility have included observations of drainage, water content, matric potential, temperature, and vegetation. Drainage is measured for all lysimeters by collecting and weighing the free water from the outlet at the base of each lysimeter.

Since 1987, 12 tests and 24 treatments have been set up in the FLTF to evaluate various combinations of soil type and layering, vegetation, and precipitation. Table A.1 summarizes all of the tests and treatments per each test. When testing was completed for some of the lysimeters, those lysimeters were converted to new tests and treatments. The tests are described below. More details about the construction and data collection, as well as a more extensive review and discussion of the data, can be found in Gee et al. (1989a), Campbell et al. (1990), Campbell and Gee (1990), Gee et al. (1993), Fayer et al. (1999), Fayer and Szecsody (2004), and Fayer and Gee (2006).

Hanford Barrier. The objective of this test was to document the performance of a Hanford Barrier. The basic configuration consisted of 1.5 m of silt loam that rested on a sequence of materials grading from sand to gravel filter layers and finally to basalt riprap. This test included comparisons of shrub-steppe vegetation and no vegetation. The treatment numbers are 1 to 4, and 7 in Table A.1.

Table A.1. Summary of Treatments and Applicable Dates at the Field Lysimeter Test Facility (after Fayer and Szecsody 2004)

Test Description	Treatment ID No.	Precipitation			Vegetation			Lysimeter ID	Monitoring Period	
		1x	2/3x	3x	NV	SRV	DRV		Start	End
Hanford Barrier	1	X					X	D4	4 Nov 1987	22 Apr 1994
		X					X	D7	4 Nov 1987	22 Apr 1994
		X					X	W1	4 Nov 1987	30 June 2007
		X					X[g]	C3	9 Nov 1988	30 June 2007
	2	X			X			D1	4 Nov 1987	30 June 2007
		X			X			D8	4 Nov 1987	27 Feb 1998
		X			X			W2	4 Nov 1987	31 Oct 1997
	3		X				X	D13	4 Nov 1987	27 Feb 1998
			X				X	D14	4 Nov 1987	22 Apr 1994
			X				X	W3	4 Nov 1987	30 June 2007
			X				X	C6	9 Nov 1988	30 June 2007
	4		X		X			D10	4 Nov 1987	8 Apr 2002
			X		X			D12	4 Nov 1987	31 Oct 1997
			X		X			W4	4 Nov 1987	31 Oct 1997
	7		X[a]		X			D9	4 Nov 1987	22 Apr 1994
			X[a]		X			D11	4 Nov 1987	22 Apr 1994
Hanford Barrier w/Gravel Admix	5	X					X	D2	4 Nov 1987	22 Apr 1994
		X					X[g]	D5	4 Nov 1987	31 Oct 1997
Eroded Hanford Barrier	6	X					X	D3	4 Nov 1987	30 June 2007
		X					X	D6	4 Nov 1987	27 Feb 1998
	18			X			X	D13	27 May 1998	30 June 2007
Gravel Mulch	8	X			X			C1	17 Nov 1989	30 June 2007
	10		X		X			C4	17 Nov 1989	30 June 2007
Pit-Run Sand	9	X					X[g]	C2	17 Nov 1989	30 June 2007
	11		X				X	C5	17 Nov 1989	30 June 2007
Basalt Side Slope	12	X			X			D2	Nov 1994	30 June 2007
	13			X	X			D9	Nov 1994	Nov 1998
Sandy Gravel Side Slope	14	X			X			D4	Nov 1994	30 June 2007
	15			X	X			D11	Nov 1994	27 Sep 2001
Prototype Barrier	16	X					X	D7	Nov 1994	Nov 1998
	17			X			X	D14	Nov 1994	31 Aug 2002
Hanford Barrier Erosion/Dune Sand Deposition	19	X				X		D5	17 Nov 1997	30 June 2007
		X				X		W2	17 Nov 1997	30 June 2007
	20			X		X		D12	17 Nov 1997	30 June 2007
				X		X		W4	17 Nov 1997	30 June 2007
Sand Dune Migration	21	X				X		D6	22 Jul 1998	30 June 2007
	22			X		X		D8	22 Jul 1998	30 June 2007
Modified RCRA Subtitle C Barrier	23	X					X	D7	23 Feb 1999	30 June 2007
	24			X			X	D9	23 Feb 1999	30 June 2007

Vegetation Symbols: NV = no vegetation, SRV = shallow-rooted vegetation, and DRV = deep-rooted vegetation.
Superscripts: "a" = irrigation accelerated until drainage commenced; "g" = sagebrush planted but died, leaving only grasses.
Note: The yellow shading indicates the current set of tests.

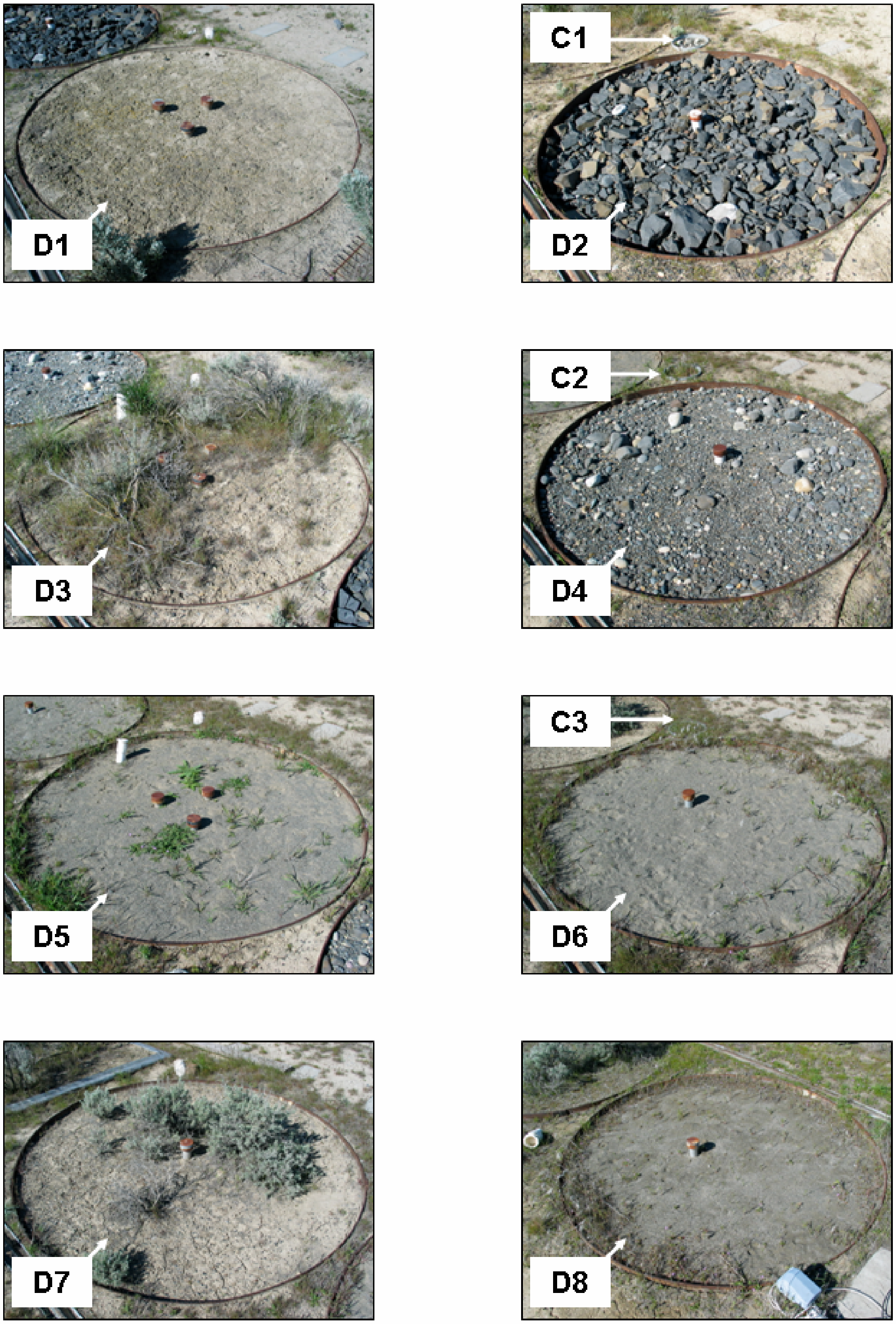

Figure A.3. Surface Conditions of the FLTF Lysimeters on 19 April 2007 (1 of 3). Drainage, weighing, and clear-tube lysimeters are labeled D, W, and C, respectively, in the photos. For scale, the exposed rim of the circular drainage lysimeters is 2 m in diameter.

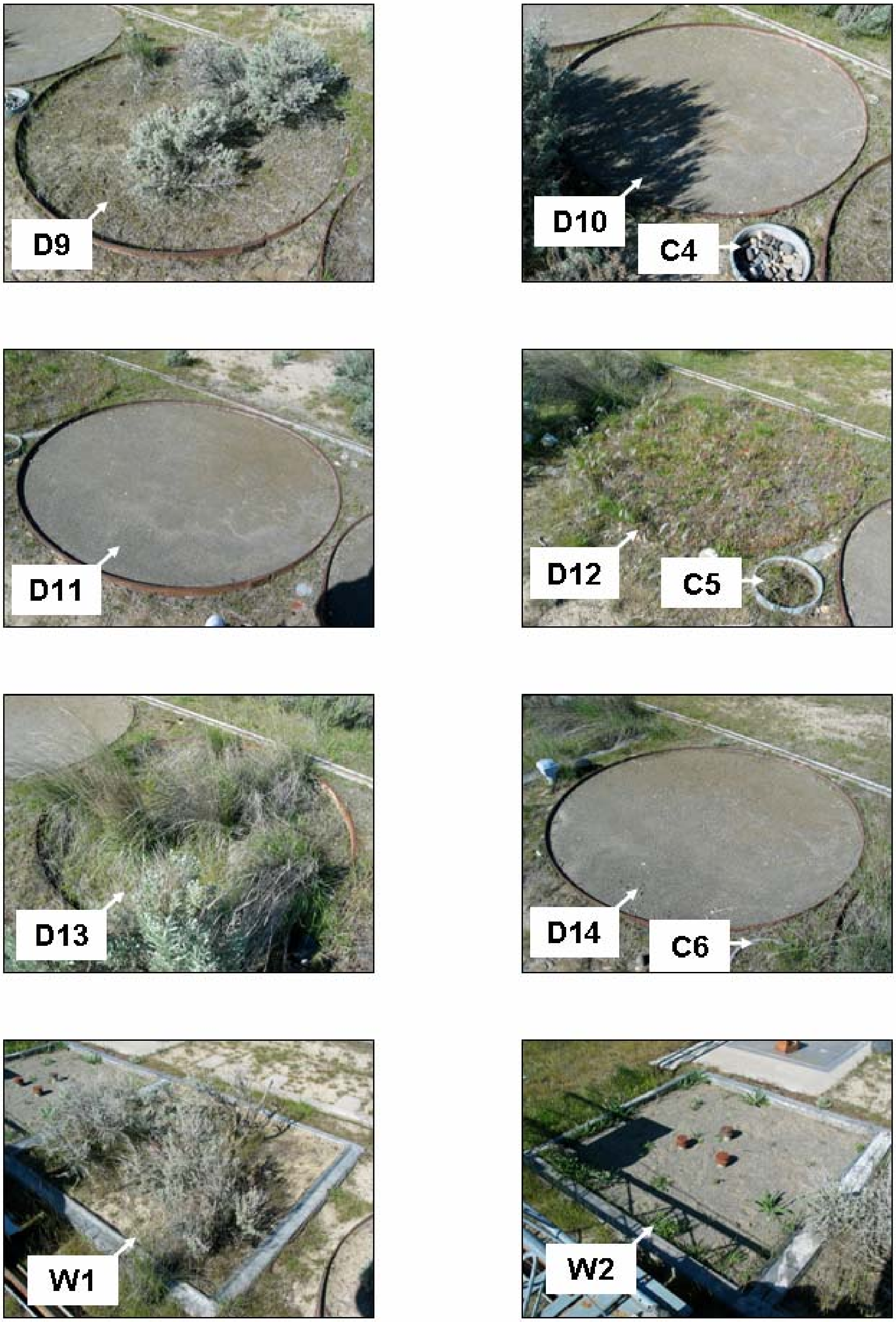

Figure A.3. (contd) (2 of 3)

Figure A.3. (contd) (3 of 3)

Hanford Barrier with Gravel Admix. The objective of this test was to document the impact of a gravel admix on the performance of a Hanford Barrier. The basic configuration was a Hanford Barrier, with the exception that the top 0.2 m of silt loam was amended with pea gravel to protect against possible erosion. The gravel content was 15% by weight. This test included shrub-steppe vegetation but addressed only ambient precipitation. The treatment number was 5; this test was terminated in 1997.

Eroded Hanford Barrier. The objective of this test was to document the performance of an eroded Hanford Barrier. The basic configuration was a Hanford Barrier design, except that the silt loam layer thickness was reduced from 1.5 to 1.0 m. In many respects, this design is similar to that of the Modified RCRA Subtitle C Barrier (RCRA Barrier) (see below). This test included shrub-steppe vegetation. The treatment numbers are 6 and 18.

Gravel Mulch. The objective of this test was to document the performance of a gravel mulch layer above Hanford formation sand. The basic configuration was 0.15 m of clean coarse gravel (no fines) above 1.35 m of *pit-run* sand (sand taken from a nearby borrow pit) and screened to remove the gravel, on top of unscreened pit-run sand (described below). This test was conducted only in the clear-tube lysimeters. The test did not include vegetation. The treatment numbers are 8 and 10. Although not its primary purpose, this test may be useful for characterizing deep drainage rates at the tank farms at Hanford. However, the results would represent an extreme and unlikely scenario because the clean coarse gravel with no fines would create an unstable surface for tank farm operations personnel while at the same time minimizing evaporation rates and optimizing drainage.

Pit-Run Sand. The objective of this test was to document the performance of a coarse gravelly sand taken from the nearby borrow pit. The basic configuration was 1.5 m of screened pit-run sand (to remove the gravel) on top of unscreened pit-run sand. This test was conducted only in the clear-tube lysimeters. The test included shrub-steppe vegetation initially, but the shrub eventually died and only shallow-rooted plants such as cheatgrass and Sandberg's bluegrass remained. The treatment numbers are 9 and 11.

Basalt Side Slope. The objective of this test was to document the performance of basalt riprap that could be used to construct side slopes for surface barriers. The basic configuration was 1.5 m of unscreened basalt riprap. This material is being tested for side slope use on a larger scale at the prototype barrier in the 200-BP-1 Operable Unit (Ward and Gee 1997). Beneath the basalt layer was a 0.15-m-thick asphaltic concrete layer underlain by gravel and more basalt riprap. Resting on top of the asphaltic concrete were about 2 to 3 cm of silt loam, within which was embedded a 2.54-cm-outside-diameter fiberglass wick. The wick was splayed within the silt loam to maximize contact but exited through the drain outlet as one piece. This test did not include vegetation. The treatment numbers are 12 and 13.

Sandy Gravel Side Slope. The objective of this test was to document the performance of unprocessed local sandy gravel that could be used to construct side slopes for surface barriers. The basic configuration was 1.5 m of sandy gravel resting on an asphaltic concrete layer in a manner similar to the basalt side slope test. The sandy gravel material was tested for side slope use on a larger scale at the prototype barrier, where it is called clean-fill gravel (Ward and Gee 1997). This test did not include vegetation. The treatment numbers are 14 and 15. Although not its primary purpose, this test may be useful for characterizing deep drainage rates at the high-level waste tank farms at Hanford that have similar textures.

Prototype Barrier. The objective of this test was to document the performance of the prototype Hanford Barrier (prototype barrier) (Ward and Gee 1997). The basic configuration was 1 m of silt loam amended with pea gravel (15% by weight) above 1 m of silt loam, which gave a combined thickness of 2 m. Beneath the silt layer were sand and gravel filter layers, then the asphaltic concrete layer described in the basalt side slope test description. Drainage was measured both above and below the asphalt layer. The test at the FLTF included shrub-steppe vegetation. The treatment numbers were 16 and 17. This test was terminated in 2002.

Hanford Barrier Erosion/Dune Sand Deposition. The objective of this test was to document the performance of the Hanford Barrier after experiencing some erosion of the silt loam layer and subsequent deposition of dune sand. The top 20 cm of silt loam were removed from four lysimeters containing a Hanford Barrier. The excavated silt loam was replaced with dune sand obtained from the dune aligned along the southern edge of the IDF site (Reidel 2005). This test included shallow-rooted vegetation, primarily cheatgrass; deep-rooted vegetation was kept off to mimic what might happen if fire prevented establishment of shrubs. The treatment numbers are 19 and 20.

Sand Dune Migration. The objective of this test was to document the performance of a sand dune that might migrate into the vicinity of or onto the surface barrier. The basic configuration was 3 m of dune sand obtained from the dune aligned along the southern edge of the IDF site (Reidel 2004). This test included shallow-rooted vegetation, primarily cheatgrass; deep-rooted vegetation was kept off to mimic what might happen if fire prevented establishment of shrubs. The treatment numbers are 21 and 22.

Modified RCRA Subtitle C Barrier. The objective of this test was to document the performance of the RCRA barrier design (DOE 1999). This barrier design meets the requirements for a RCRA barrier using only 1 m of silt loam rather than the 2 m of silt loam used for the prototype barrier design. In addition, the silt layer has two modifications. First, the upper 0.5 m of silt loam was amended with pea gravel at the rate of 15% by weight for erosion protection. Second, the lower 0.5 m of silt was compacted to create a low-conductivity layer to impede downward drainage (DOE 1999). Construction of this test required slight design modifications to the sand filter layer, the gravel drainage layer, and the density of the compacted silt layer (Fayer et al. 1999). The layers reside on an asphaltic concrete layer; drainage is measured both above and below the asphalt layer. This test included shrub-steppe vegetation. The treatment numbers are 23 and 24.

For each test, average drainage rates for each previously described treatment are provided in Table A.2 (Fayer and Szecsody 2004; Fayer and Gee 2006). Fayer and Gee (2006) describe that pattern of drainage from the lysimeters, where it exists, to be generally predictable, with drainage occurring in the

Table A.2. Average Drainage Rates for Selected Periods at the Field Lysimeter Test Facility (Sources: Fayer and Szecsody 2004; Fayer and Gee 2006)

Test Description	Treatment ID No.	Lysimeter ID	Averaging Period Start	Averaging Period End	Averaging Period Duration (yr)	Average Drainage (mm/yr)
Hanford Barrier	1	D4	2 Jan 1990	19 Apr 1994	4.3	0.0
		D7	2 Jan 1990	19 Apr 1994	4.3	0.0
		W1	4 Nov 1987	31 Mar 2004	16.4	0.0[(a)]
		C3	9 Nov 1988	31 Mar 2004	15.4	0.0
	2	D1	3 Jan 1991	31 Mar 2004	13.2	0.0
		D8	2 Jan 1990	25 Feb 1998	8.2	0.2
		W2	4 Nov 1987	31 Oct 1997	10.0	0.0
	3	D13	2 Jan 1990	7 Jan 1998	8.0	0.0
		D14	2 Jan 1990	5 Jan 1994	4.0	0.0
		W3	2 Jan 1990	31 Mar 2004	14.3	0.0[(a)]
		C6	2 Jan 1990	31 Mar 2004	14.3	0.0
	4	D10	2 Jan 1990	10 Jan 2002	12.0	10.7
		D12	2 Jan 1990	31 Oct 1997	7.8	16.4
		W4	2 Jan 1990	31 Oct 1997	7.8	6.2
Hanford Barrier w/Gravel Admix	5	D2	2 Jan 1990	19 Apr 1994	4.3	0.0
		D5	2 Jan 1990	31 Oct 1997	7.8	0.0
Eroded Hanford Barrier	6	D3	2 Jan 1990	31 Mar 2004	14.3	0.0
		D6	2 Jan 1990	25 Feb 1998	8.2	0.0
	18	D13	27 May 1998	31 Mar 2004	5.8	2.4
Gravel Mulch	8	C1	2 Jan 1990	30 Dec 2003	14.0	89.0
	10	C4	2 Jan 1990	30 Dec 2003	14.0	333
Pit-Run Sand	9	C2	2 Jan 1990	30 Dec 2003	14.0	25.1
	11	C5	2 Jan 1990	30 Dec 2003	14.0	79.9
Basalt Side Slope	12	D2	4 Jan 1995	30 Dec 2003	9.0	58.9
	13	D9	4 Jan 1995	24 Nov 1998	3.9	269
Sandy Gravel Side Slope	14	D4	4 Jan 1995	30 Dec 2003	9.0	113
	15	D11	4 Jan 1995	Sep 2001	6.8	365
Prototype Barrier	16	D7	4 Jan 1995	24 Nov 1998	3.9	0.0
	17	D14	4 Jan 1995	28 Aug 2002	7.7	0.0
Hanford Barrier Erosion/Dune Sand Deposition	19	D5	17 Nov 1997	31 Mar 2004	6.4	0.14
		W2	17 Nov 1997	31 Mar 2004	6.4	0.0
	20	D12	17 Nov 1997	18 Nov 2003	6.0	139
		W4	17 Nov 1997	18 Nov 2003	6.0	66.8
Sand Dune Migration	21	D6	26 May 1999	18 Apr 2007	7.8	14.2[(b)]
	22	D8	26 May 1999	31 Mar 2004	4.8	234
Modified RCRA Subtitle C Barrier	23	D7	23 Feb 1999	31 Mar 2004	5.1	0.0
	24	D9	23 Feb 1999	31 Mar 2004	5.1	0.04

(a) Drainage rate may be affected by bottom boundary.
(b) Data collected after publication of Fayer and Szecsody (2004) were included to provide a better estimate of the average drainage rate.

Note: The yellow shading indicates the current set of tests. Data collected after the end of the averaging period have not yet been published.

Precipitation symbols: ambient = natural precipitation; 2x/3x = ambient precipitation plus irrigation so that total water received is equivalent to 2x (or 3x) the average annual precipitation (considered to be 160 mm/yr); for some tests, the precipitation treatment started as 2x and was switched several years later to 3x.

spring and early summer in response to winter precipitation. In the Hanford Barrier test, two lysimeters (W1 and W3) had sagebrush and relatively shallow (1.7 m) bottom boundaries that may have contributed to the observation of zero drainage. Several other lysimeters (D4, D7, D13, and D14) with sagebrush and deeper bottom boundaries yielded zero drainage as well, suggesting that the lower boundary in W1 and W3 did not have a significant influence on drainage.

Fayer and Szecsody (2004) report that the sand dune migration lysimeter receiving ambient precipitation (D6) first experienced measurable drainage in March 2004, the last month of the reported drainage record. Because the drainage record included only a single month of drainage, the authors stated that the average drainage rate of 1.2 mm/yr was a preliminary drainage rate. Although not reported elsewhere, drainage from this lysimeter has continued since March 2004, averaging 14.2 mm/yr through 18 April 2007.

A.2 200 East Lysimeter

The 200 East Lysimeter site is approximately 1.6 km south of the 200 East Area. The site includes a closed bottom lysimeter (18.5 m deep and 3 m in diameter) and a companion open bottom lysimeter constructed to investigate recharge conditions on the Central Plateau. A schematic of the 200 East closed bottom lysimeter is presented in Figure A.4. At the time of construction in 1971, both lysimeters were instrumented with thermocouple phychrometers, temperature sensors, and neutron probe access tubes (Hsieh et al. 1973). The lysimeters were filled with homogenized sand to loamy sand material excavated from the site. The only monitoring done at this site has been for water content using the neutron probe ports (Hsieh et al. 1973; Jones 1978; Gee et al. 1994). The other instrumentation was lost during installation and was never replaced.

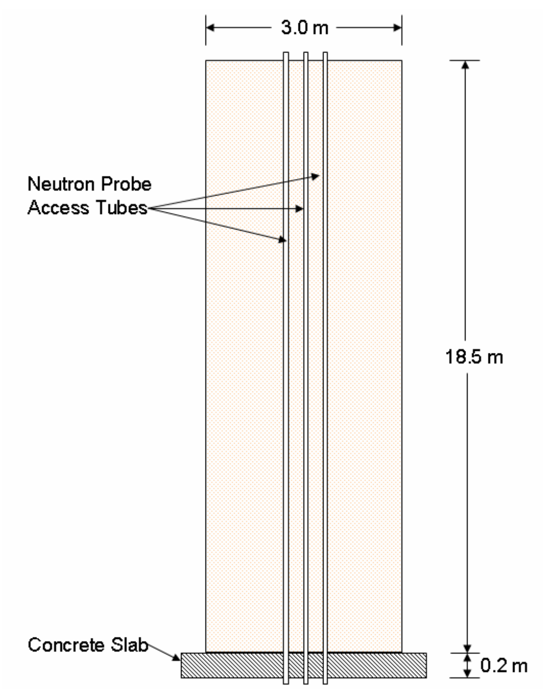

Figure A.4. Cross Section of the Closed-Bottom 200 East Lysimeter

Estimates of recharge based on 200 East Lysimeter observations have varied. Jones (1978) proposed that recharge conditions could be as high as 5 mm/yr based on the assumption of unit gradient conditions, assumed unsaturated hydraulic conductivity, and measured moisture content within the closed-bottom lysimeter.

Routson and Johnson (1988) estimated recharge by examining the change in moisture content of the closed-bottom lysimeter sediment from the time the lysimeter was packed in 1971 to 1985. Over this time interval, the lysimeter surface contained tumbleweed and shallow-rooted grasses. The authors observed that the difference in moisture content measured in 1985 did not differ significantly from that measured in 1971, and they concluded recharge to be equal to 0 ± measurement error of 2 mm/yr.

Gee et al. (1994) reported results from a neutron logging of water contents in the closed-bottom 200 East Lysimeter between 1988 and 1991. Until

March 1991, the lysimeter was kept bare of vegetation to observe whether deep infiltration would occur in the lysimeter without the presence of vegetation that could extract water from the soil. The measurements revealed that water content in the lysimeter increased to a depth of 3 m. Once vegetation became reestablished after March 1991, water content quickly decreased to pre-1988 values. Although a quantitative estimate of drainage was not made, the results showed that, with plants on the surface, the soil in the lysimeter could limit recharge to values too small to detect in 14 years. The results also showed that, without plants, the soil could not limit recharge through evaporation alone and that deep drainage rates were significant enough to be detected within the 3-year monitoring period.

A.3 200 East Deep Well Test Site

Within 30 m of the 200 East Lysimeter site, four wells were installed between 1969 and 1970 for performing recharge and water balance studies. Hsieh et al. (1973) describe the vegetation at the 200 East Deep Well Test Site to be predominantly sagebrush and cheatgrass. Soil at this location is a sand to loamy sand containing small amounts of gravel. Two of these wells, wells 32-49C and 32-49D, were instrumented from near the ground surface to the water table at 95 m with thermocouple psychrometers and temperature transducers before being backfilled. The instruments were installed to measure temperature and matric potential gradients in the field. Well 32-49C was hand-augered and instrumented to a depth of approximately 4.4 m (Brownell 1971; Enfield and Hsieh 1971). Well 32-49D was drilled to the water table at 95 m and instrumented at intervals along the entire depth of the borehole (Hsieh et al. 1973).

Analysis by Enfield and Hsieh (1971) of the well 32-49C thermocouple psychrometer data indicates that an upward flux of water exists above the 3 m depth, accompanied by a downward flux below 3 m. Using the thermocouple psychrometer and temperature transducer data from well 32-49D, as well as estimates of soil hydraulic and thermal properties, Enfield et al. (1973) estimated the flux due to gravitational and matric potential gradients was 0.3 mm/yr in the downward direction. The estimated flux due to the thermal gradient was 0.04 mm/yr in the upward direction, producing a net downward flux of 0.26 mm/yr. Enfield et al. (1973) cautions about the preciseness of their estimate, given the uncertainties in estimated soil hydraulic and thermal properties as well as not considering the effects of osmotic potentials.

A.4 Prototype Hanford Barrier

The Prototype Hanford Barrier is a full-scale capillary barrier constructed over a decommissioned waste disposal site (216-B-57 crib) in the 200 East Area. Figure A.5 shows the position of the barrier relative to the nearby SST BY farm. The barrier was constructed in 1994 to evaluate the field-scale physical and hydrologic performance, as well as constructability and cost, of the Hanford Barrier. The barrier is composed of a 1.0-m-thick silt loam/pea gravel admix surface layer followed by a 1.0-m-thick silt loam layer, thin sand and gravel filter layers, and a 1.5-m-thick basalt riprap layer underlined by an asphalt pad. Two different side slope configurations exist on the Prototype Hanford Barrier. One is a 2:1 slope made up of basalt riprap, while the other is a 10:1 slope of coarse sandy gravel material. A healthy stand of shrubs and grasses exists on the surface of the barrier. The sandy gravel side slope contains grasses and a sparse distribution of shrubs that have become increasingly established since construction of the barrier. Figure A.6 shows the condition of the sandy gravel side slope plant community in 2004. The riprap side slope is devoid of vegetation.

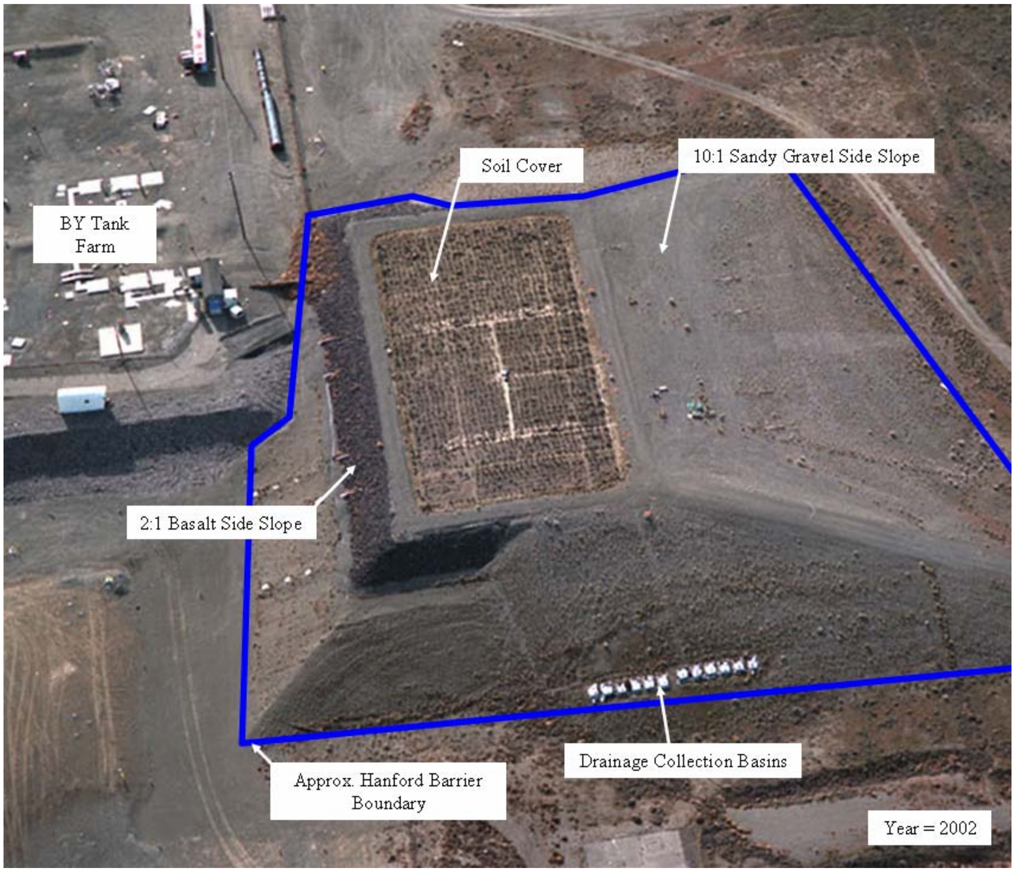

Figure A.5. Aerial View of the Prototype Hanford Barrier Looking in Southerly Direction. Image captured in 2002.

Figure A.6. Prototype Hanford Barrier Sandy Gravel Side Slope in 2004. The view is looking up the side slope and to the southeast. SST BY farm is just over the horizon. Plant density on the lower slopes is far higher than near the top where the side slope drainage pads are located.

The asphalt pad beneath the barrier is subdivided into twelve rectangular bermed plots. Eight of the twelve plots are larger (23 by 14 m; area = 322 m^2) and used for drainage measurements. Figure A.7 shows the surface designation of each plot. Drainage water that reaches the asphalt pad is diverted to a collection basin where it is measured using a combination of tipping bucket measurements and water height within the basin, determined by monitoring the hydrostatic pressure within the basin. Figure A.8 shows the outline of the drainage pads projected on an aerial view the barrier in 2002. Note the limited vegetation above drainage pad 1W.

In addition to drainage, long-term performance monitoring of the Prototype Hanford Barrier consists of precipitation and soil moisture measurements. DOE (1999) and Wittreich et al. (2003) describe the barrier monitoring equipment and construction in more detail.

The surface above drainage plots 1W and 4W is primarily the coarse sandy gravel side slope material, but it also includes the gravel access road and a small portion of the silt loam cover. The surface above plots 1E and 4E is primarily the basalt side slope as well as the gravel access road and a small portion of the silt loam cover. The remaining plots are covered with silt loam soil. The northern plots 4W, 4E, 5W, 5E, 6W, and 6E all were irrigated. Soil plots 5W, 5E, 6W, and 6E were irrigated for a 3-year period from 1994 to 1997 to simulate increased rainfall. Side slope plots 4E and 4W were irrigated for a 1-year period from 1996 to 1997.

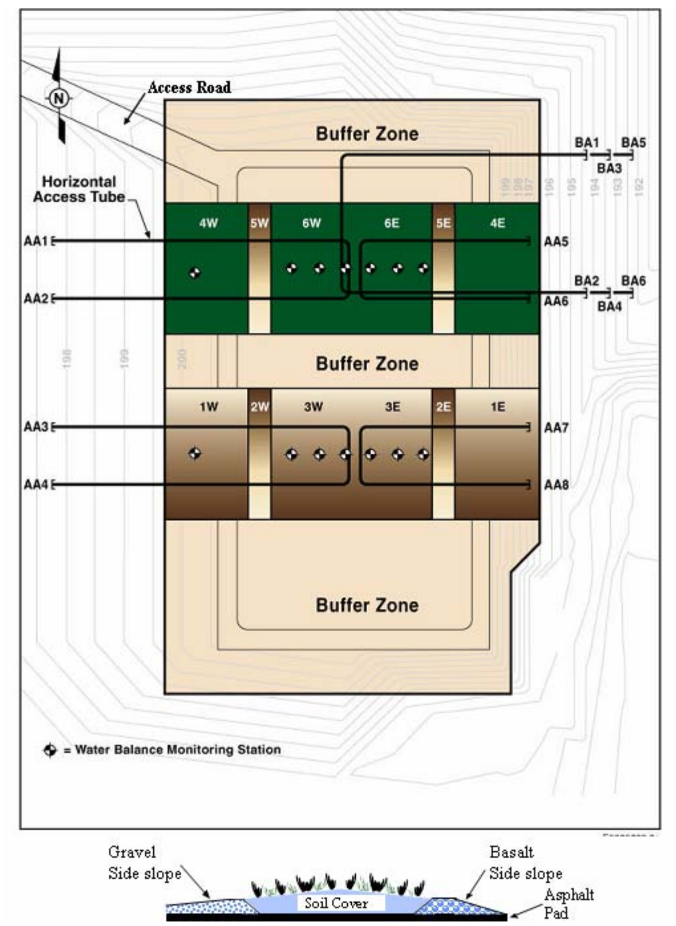

Figure A.7. Plan and Side Schematic of the Prototype Hanford Barrier Including the Drainage Plot Locations, Water Balance Monitoring Stations, and Horizontal Neutron Probe Access Tubes. Green colored plots received irrigation to increase total water received to 3x the average annual amount. Brown colored plots received only ambient precipitation. For scale, the maximum width of the colored region is 64 m. (Source: Gee et al. 2002)

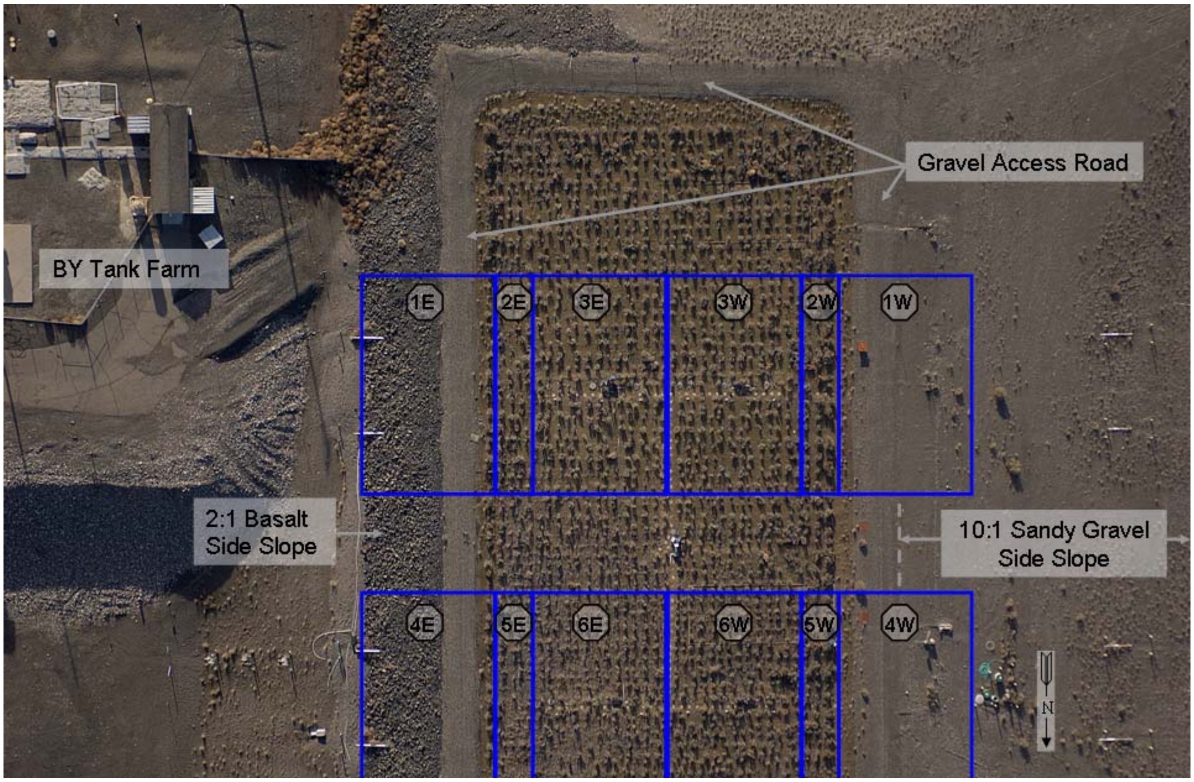

Figure A.8. Outline of Drainage Pads Projected on Barrier Surface in Aerial Photo from 2002 (photo provided courtesy of Andy Ward, PNNL)

Ward et al. (2005) reported drainage measurement results for the full-sized plots during the monitoring period October 1994 through September 2004. Measurements show that during and after the irrigation period, all irrigated plots had increased drainage relative to the nonirrigated plots. At the onset of drainage monitoring, vegetation on the silt loam surface and sandy gravel side slope had yet to establish. At the same time, from 1995 to 1997, annual precipitation at the Prototype Hanford Barrier was the highest measured during the 10-year record. Thus, the drainage rates from the 10-year record represent rates that incorporate an emerging plant community during a time of higher-than-average precipitation. For the nonirrigated basalt plot (1E), the 10-year monitoring period produced an average drainage of 30.2 mm/yr. This drainage value is 15% of the 10-year average annual precipitation of 200.4 mm/yr. For the nonirrigated sandy gravel side slope plot (1W), Ward et al. (2005) reported drainage to be 38.6 mm/yr or 19% of the cumulative precipitation for that time. The nonirrigated silt loam soil cover plots were reported to have an average drainage of essentially zero for plot 3W and 0.009 mm/yr for plot 3E. Table A.3 summarizes the nonirrigated plots average drainage rates and the percentage of annual average precipitation represented by the drainage rates. The presence of the silt loam cover and gravel road over a portion of plots 1W and 1E complicates interpretation of drainage measurements in regard to the relationship to a single surface condition. The silt loam cover would be expected to minimize drainage compared to sandy gravel and basalt, whereas the gravel roadway, due to its coarse texture and lack of vegetation, would likely produce increased drainage relative to the primary surface treatment. However, sandy gravel (1W) and basalt (1E) predominantly make up the material of their respective plots; thus, it is expected that these materials dominate the drainage behavior of the side slope plots.

Table A.3. Prototype Hanford Barrier Average Drainage Rates from Nonirrigated Plots (Ward et al. 2005)

Average Precipitation (mm/yr)	Average Drainage from Nonirrigated Plots (mm/yr [percentage of precipitation received])			
	Sandy Gravel (1W)	Silt Loam (3W)	Silt Loam (3E)	Basalt (1E)
200.4	38.6 [19.3%]	2.9E-5 [1.5E-5%]	0.009 [0.004%]	30.2 [15.1%]

A.5 Solid Waste Landfill

The Hanford Solid Waste Landfill (SWL) is in the 600 Area approximately 6.5 km northwest of the Wye barricade. A basin lysimeter was installed at the SWL in 1992 to assess the quality and quantity of leachate leaving the landfill to satisfy regulatory requirements (DynCorp 2000; Wittreich and Wilson 1991); monitoring of the basin lysimeter continues for this purpose. The lysimeter was built using a high density polyethylene plastic liner to create a basin-shaped collection pad (much like the drain pads of the Prototype Hanford Barrier) with a capture area of 85 m^2 at a depth of approximately 6.5 m. Like other basin lysimeters, the SWL has no side walls. The lysimeter was subsequently filled with nonorganic solid waste mixed with Hanford formation sands and gravels. Once filled, the surface of the landfill was finished with 0.5 m of sandy-gravel backfill material. Drainage water intercepted by the basin lysimeter is routed to a collection sump, where it is pumped to collection vessels on the surface. At regular intervals, the volume of water collected is measured and leachate water analyzed for organic and nonorganic chemicals. Additional information about the basin lysimeter can be found in DynCorp (2000) and Wittreich and Wilson (1991).

Over time, vegetation in the form of non-native species, including Indian wild rice (bunchgrass), has taken root at the site (Gee et al. 2005a). However, vehicle traffic passes directly over approximately 10% of the basin lysimeter, resulting in no vegetation in this area. The surface conditions above the location of the basin lysimeter are shown in Figure A.9.

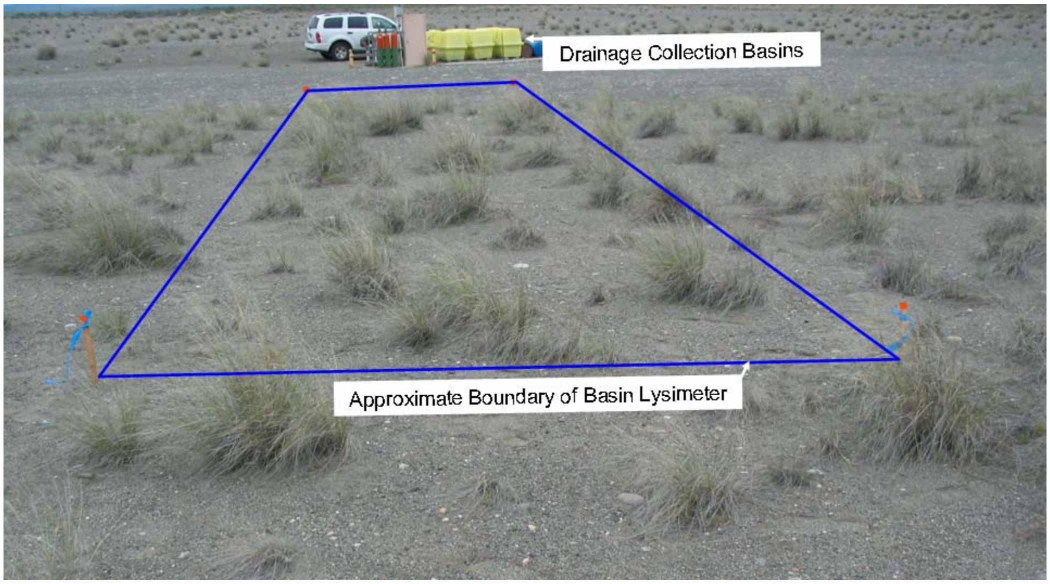

Figure A.9. Approximate Boundary of the Basin Lysimeter at the Hanford Solid Waste Landfill

Measurable drainage by the basin lysimeter first occurred in July 1996 and has continued uninterrupted since. DOE reports the most current basin lysimeter drainage results, spanning nearly 10 years, from July 1996 to October 2006.[a] The measured drainage over that time yields an average drainage of 51.1 mm yr^{-1}. This equates to 29.2% of the precipitation recorded at the Hanford Meteorological Station during the same period.

A.6 300 North Lysimeter Facility

The 300 North Lysimeter Facility, formerly known as the Vadose Zone Test Facility (VZTF), and previously called the Buried Waste Test Facility (BWTF), is about 10 km north of Richland. The lysimeters were constructed originally to support water balance and radionuclide transport studies, with water balance monitoring continuing to date. At this location, a suite of lysimeters was installed in 1978. A schematic of the 300 North Lysimeter Facility is provided in Figure A.10. Two of the lysimeters are called the North Caisson and South Caisson. Completed in 1979, these two lysimeters are both 2.7 m in diameter and 7.6 m deep and filled with sand screened from excavated material used to construct the site. A schematic of the 300 North Lysimeter Facility is provided in Figure A.10. The screened material is Hanford formation material screened to

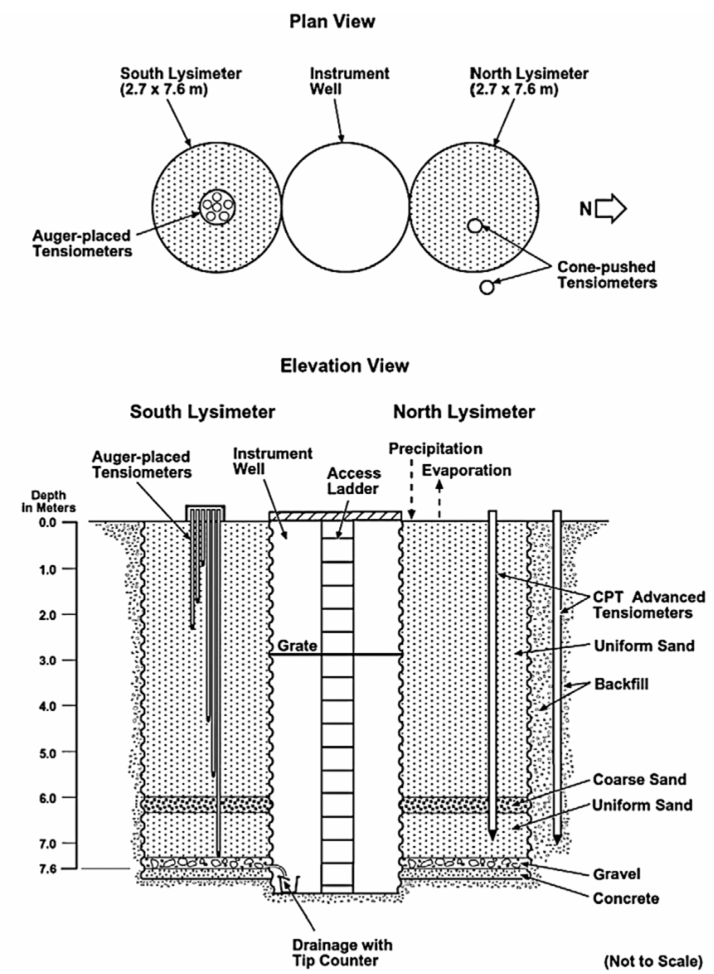

Figure A.10. Plan View and Cross-Section View of 300 North Lysimeter Facility

(a) U.S. Department of Energy. 2006. *Hanford Site Solid Waste Landfill Annual Monitoring Report, July 2005 through June 2006*. Letter report to Washington State Department of Ecology.

contain less than 1% gravel (material > 2 mm). Additional details about the construction of the facility can be found in Phillips et al. (1979). In 1999, the lysimeters were instrumented with tensiometers and moisture content sensors at various depths. Nearly continuous automated monitoring of soil-water pressure and soil-water content has continued in the South Caisson. While not near the Central Plateau, the 300 North lysimeter soil and surface conditions are similar to conditions found at disturbed locations on the Central Plateau.

Over the life of the lysimeters, the surface has remained essentially devoid of vegetation. Surface conditions of the South Caisson lysimeter are shown in Figure A.11.

Drainage out of the base of the South Caisson lysimeter began in 1981 and since that time has been measured in one of two ways. From the inception of drainage in 1981 until April 2000, drainage was measured by periodically collecting water in tared containers, returning the containers to the laboratory where they were manually weighed on an electronic balance. In April 2000 and thereafter, the drainage collection was switched to an automatic sampling scheme using a tipping spoon rain gauge. Gee et al. (2005b) compiled the South Caisson drainage data from 1981 to 2005, reporting a drainage rate of 62 mm/yr.

Figure A.11. Surface of the South Caisson Lysimeter at the 300 North Lysimeter Facility. The capped PVC pipe is the housing for the lysimeter tensiometer nest.

A.7 Tank Farm Monitoring

WMAs B, TX, and T have been equipped with vadose zone monitoring instrumentation to quantify recharge conditions beneath WMA surfaces. Vadose zone sensors were installed in WMA B in July 2001 to measure soil water pressure, soil temperature, soil water content, and soil water flux and automated measurements began immediately. Sensors were installed in WMA TX in October 2002 to measure soil water pressure and temperature. In addition, two soil water flux measurement devices were installed just outside the fence around WMA TX. Gee et al. (2003) described the sensor installation in both farms and a subset of the data collected up to early November 2002. Automated data collection has continued since 2002, but unpublished analysis suggests that a number of the instruments have failed or are not functioning correctly, making use of the data difficult.[a]

A series of soil water samples were taken at the B tank farm using manually-operated solution-sampler instrumentation (Serne et al 2002). The water sample data are of interest in transport analysis and indirectly relates to drainage estimation. The manual instrumentation appears to work well for repeatedly removing water samples from great depth (more than 70 m). The data indicate that solution chemistry is disturbed by the borehole placement and may take several years to equilibrate, but could prove useful in estimating plume migration and recharge rates.

In September 2006, vadose zone monitoring instruments were installed in the T Tank Farm in support of a demonstration interim surface barrier. The installed instruments include sensors for measuring soil water pressure, soil temperature, soil water content, and soil water flux. Information about the installation and monitoring plan are available in Zhang et al. (2007). A cursory review of the soil water pressure and moisture content results suggests that moisture from winter precipitation has reached a depth of 2 m below ground surface by 15 May 2007, despite 2006–2007 winter precipitation being near average. Results from the T Tank Farm vadose zone sensors are expected to be analyzed and published by the end of 2007.

In addition to instrument nests, ongoing efforts continue to measure soil moisture content and radionuclide contaminant distributions using various geophysical methods as well as spectral gamma and neutron logging (e.g., Wood and Jones 2003). Soil temperature profiles near tanks also are monitored in some instances. While the focus of monitoring is not to produce recharge estimates, the potential exists to estimate recharge from these measurements. With reliable knowledge of the hydraulic properties of sediments beneath the tank farms, measured moisture contents could be used to estimate recharge rates. Monitoring the rate of movement of contaminant plumes over time also provides a means to estimate recharge rates below tank farms. While a comprehensive attempt to estimate recharge using the aforementioned methods has not been made, the potential exists.

A.8 Integrated Disposal Facility

A field site was established in 2000 to evaluate the potential for recharge in the sand dune along the southern edge of the Integrated Disposal Facility (IDF) in the 200 East Area. Multiple sensors were installed along with 13 access tubes that allowed measurement of soil water content using a neutron probe. The access tube distribution provided three treatments: mature shrub (6 access tubes), juvenile

(a) Readers interested in the data collected from sensors in the WMAs B, TX, and T should contact Dave Myers, CH2M HILL, at 509-373-3972 or David_A_Dave_Myers@rl.gov shrubs (4 access tubes), and intershrub (3 access tubes). The details of the sensor and access tube installation were described by Kirkham and Kahn in 2000.[a] Starting in 2000, neutron moisture probe measurements have been made at the IDF site a minimum of two times per year to capture the seasonal water dynamics beneath the three treatments. To date, the IDF neutron probe monitoring data have not been published.

A.9 Grass Site

Located approximately 4.5 km northwest of the 300 Area, the Grass Site was established in 1983 to study recharge conditions. The site contained 25 neutron probe access tubes spaced 6 m apart in a 5 × 5 grid. At this location, a layered soil exists, with a sandy loam to loamy sand soil present from the surface to approximately 40 cm and a sandy soil classified as a Rupert sand beneath that (Gee et al. 1989b). The vegetative cover comprises annual and perennial grasses and is devoid of deep-rooted vegetation such as sagebrush. More recent activities at this site include monitoring drainage and soil water content using water fluxmeters and capacitance probes installed in early 2005. Results from these more recent installations have not been published.

During an 8-year period starting in 1983, soil water content conditions were measured at the Grass Site using neutron probe measurements. Results of these measurements were used in Appendix A of Fayer and Walters (1995) to estimate recharge rates from measured changes in soil-water storage. From this analysis, an 8-year average estimated recharge rate of 25.4 mm/yr was calculated. A significant finding from this analysis was that recharge estimates were observed to be spatially highly variable when comparing recharge estimates from individual access tubes. In the extreme instance, in a single year the highest estimated recharge value was 28 times the lowest estimated recharge value.

A.10 Summary of Water Balance Recharge Studies

A number of water balance studies varying in treatments (soil and vegetation), location, measurement method, measurement depth, and monitoring period have been carried out at the Hanford Site. Those studies relevant to the Central Plateau have been identified in previous sections and, if available, measured drainage rates cited. Table A.4 summarizes these drainage rates and monitoring conditions. For those studies in which monitoring is ongoing, accuracy of the estimated recharge rates will continue to improve as the measurement record grows and incorporates more climatic extremes. For all sites, a concerted effort is needed to obtain an accurate accounting of the particle-size distribution, including gravel fraction, comprising the surface soil of the monitoring location. Given the sensitivity of soil-water evaporative behavior to even small fractions of silt and clay, such particle size information will enhance the applicability of applying estimated recharge rates to specific soil types that for now are restricted to general descriptive statements such as "sandy gravel."

(a) Kirkham RR and FO Khan. 2000. *Status Report: FY 2000 Field Measurements Task for the ILAW Disposal Site*. Letter report for the IDF project.

Table A.4. Summary of Hanford Water Balance Studies for the Central Plateau

Site	Test Description	Measurement Depth (m)	Vegetation	Precipitation	Averaging Period			Average Drainage (mm/yr)
					Start	End	Duration (yr)	
Field Lysimeter Test Facility	Hanford Barrier	3	DRV	ambient	Jan 1990	Apr 1994	4.3	0
		3	DRV	ambient	Jan 1990	Apr 1994	4.3	0
		1.7	DRV	ambient	Nov 1987	Mar 2004	16.4	0
		3	DRV	ambient	Nov 1988	Mar 2004	15.4	0
		3	NV	ambient	Jan 1991	Mar 2004	13.2	0
		3	NV	ambient	Jan 1990	Feb 1998	8.2	0.2
		1.7	NV	ambient	Nov 1987	Oct 1997	10	0
		3	DRV	2x/3x ambient	Jan 1990	Jan 1998	8	0
		3	DRV	2x/3x ambient	Jan 1990	Jan 1994	4	0
		1.7	DRV	2x/3x ambient	Jan 1990	Mar 2004	14.3	0
		3	DRV	2x/3x ambient	Jan 1990	Mar 2004	14.3	0
		3	NV	2x/3x ambient	Jan 1990	Jan 2002	12	10.7
		3	NV	2x/3x ambient	Jan 1990	Oct 1997	7.8	16.4
		1.7	NV	2x/3x ambient	Jan 1990	Oct 1997	7.8	6.2
	Hanford Barrier w/Gravel Admix	3	DRV	ambient	Jan 1990	Apr 1994	4.3	0
		3	DRV	ambient	Jan 1990	Oct 1997	7.8	0
	Eroded Hanford Barrier	3	DRV	ambient	Jan 1990	Mar 2004	14.3	0
		3	DRV	ambient	Jan 1990	Feb 1998	8.2	0
		3	DRV	3x ambient	May 1998	Mar 2004	5.8	2.4
	Gravel Mulch	3	NV	ambient	Jan 1990	Dec 2003	14	89
		3	NV	2x/3x ambient	Jan 1990	Dec 2003	14	333
	Pitrun Sand	3	DRV	ambient	Jan 1990	Dec 2003	14	25.1
		3	DRV	2x/3x ambient	Jan 1990	Dec 2003	14	79.9
	Basalt Side Slope	3	NV	ambient	Jan 1995	Dec 2003	9	58.9
		3	NV	3x ambient	Jan 1995	Nov 1998	3.9	269
	Sandy Gravel Side Slope	3	NV	ambient	Jan 1995	Dec 2003	9	113
		3	NV	3x ambient	Jan 1995	Sep 2001	6.8	365
	Prototype Barrier	3	DRV	ambient	Jan 1995	Nov 1998	3.9	0
		3	DRV	3x ambient	Jan 1995	Aug 2002	7.7	0
	Hanford Barrier Erosion/Dune Sand Deposition	3	SRV	ambient	Nov 1997	Mar 2004	6.4	0.14
		1.7	SRV	ambient	Nov 1997	Mar 2004	6.4	0
		3	SRV	3x ambient	Nov 1997	Nov 2003	6	139
		1.7	SRV	3x ambient	Nov 1997	Nov 2003	6	66.8
	Sand Dune Migration	3	SRV	ambient	May 1999	Apr 2007	7.8	14.2
		3	SRV	3x ambient	May 1999	Mar 2004	4.8	234
	Modified RCRA Subtitle C Barrier	3	DRV	ambient	Feb 1999	Mar 2004	5.1	0
		3	DRV	3x ambient	Feb 1999	Mar 2004	5.1	0.04
200 East Lysimeter	Loamy Sand	18.5	SRV	ambient	1971	1985	14	0-2
200 East Deep Well	Sand/Loamy Sand	95	SRV/DRV	ambient	1970	1973	3	0.26
Prototype Hanford Barrier	Sandy Gravel Sideslope	4.25	SRV	ambient	1994	2004	10	38.6
	Silt Loam Cover	4.25	SRV/DRV	ambient	1994	2004	10	0.0045
	Basalt Side Slope	4.25	NV	ambient	1994	2004	10	30.2
Solid Waste Landfill	Sandy Gravel	6.5	SRV	ambient	1996	2006	10	51.1
300 North Lysimeter	Sand	7.6	NV	ambient	1981	2005	15	62

Vegetation symbols: NV = no vegetation; SRV = shallow-rooted vegetation; DRV = deep-rooted vegetation.
Precipitation symbols: ambient = natural precipitation; 2x/3x = ambient precipitation plus irrigation so that total water received is equivalent to 2x (or 3x) the average annual precipitation (considered to be 160 mm/yr); for some tests, the precipitation treatment started as 2x and was switched several years later to 3x.

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Appendix B

Tracer-Based Estimates of Recharge

Appendix B

Tracer-Based Estimates of Recharge

The direct measurement of recharge at the Hanford Site is complicated by the low soil-water fluxes that exist under many conditions at Hanford and the vertical extent of the vadose zone beneath the Central Plateau. Under such conditions, the environmental tracer technique offers the best means for estimating recharge for past conditions—either pre-Hanford, for which we have no data, or during the operations period (1943–present), for which we have limited data. Numerical modeling can supplement the tracer estimates, but its value is constrained by uncertainties in values of soil, vegetation, and climate parameters.

Environmental tracers have been shown to be credible tools for estimating recharge in arid climates similar to Hanford (Allison et al. 1994; Phillips 1994; Scanlon et al. 2006), and they provide a cost-effective alternative to long-term recharge measurement studies. Environmental tracers provide a time-averaged estimate of recharge and, depending on the length of record, information regarding historic recharge. The ability to obtain estimates of historic recharge is valuable in instances when knowledge of recharge under past surface conditions or climate regimes is desired. However, use of environmental tracers has two general disadvantages: it provides an indirect point estimate of recharge, and the accuracy of the method can be affected by subsurface heterogeneities, preferential flow, changes in deposition rate, mineral dissolution, and analytical errors, particularly when concentrations are low.

This appendix summarizes the data and recharge estimates from past tracer studies of the Hanford Central Plateau. In addition, characterization studies performed on cores collected from boreholes in or near Hanford tank farms were evaluated for use in estimating recharge using measured pore water chloride. Those characterization studies in which the chloride data were determined to be satisfactory for use in estimating recharge were analyzed using the chloride mass balance (CMB) method.

B.1 Tracer Methods

The environmental tracer technique uses the in situ pore water chemical composition to infer recharge rates. An assortment of environmental tracers has been used to estimate recharge at Hanford, including chloride and isotopes of chloride (^{36}Cl), hydrogen (^{2}H), oxygen (^{18}O), strontium (^{87}Sr), and uranium (^{234}U). Other tracers such as ^{3}H, ^{129}I, ^{14}C, ^{13}C, ^{79}Se, ^{99}Tc, and ^{15}N have been considered and/or used elsewhere to estimate recharge, but they have not been used at Hanford (Murphy et al. 1991). An overview of the methodology for each environmental tracer that has been used at the Hanford Site follows.

B.1.1 Chloride

Chloride is the tracer most frequently used to estimate recharge at Hanford because of its relatively inexpensive analysis cost and the simplicity with which estimates of recharge can be made from knowledge of pore water chloride concentrations. Chloride is deposited on the soil surface by precipitation and dry fallout. As the precipitation infiltrates the soil, it transports the chloride into the soil profile. Once in the soil profile, chloride is transported predominantly by liquid water. The processes of evaporation and transpiration alter the water content without materially changing the chloride content, thus inexorably increasing the concentration of chloride in the soil water relative to that in precipitation. Evaporation removes the soil water at the soil surface. Transpiration (by plants) removes water from throughout the soil profile wherever roots are located. Some chloride is taken up by plants, but the net change in pore water chloride concentration is small and generally negligible when averaged over several years.

When evaluated below the plant root zone, the increased chloride concentration can be related to the deep water flux, i.e., the recharge rate. With knowledge of the quantity of chloride introduced to the soil surface and information about the chloride profile in the subsurface, an estimation of recharge can be made using

$$J_R = \left(\frac{Cl_p}{Cl_s} \right) \times P \tag{B.1}$$

where J_R = recharge (mm/yr)
Cl_p = average chloride concentration in precipitation and dry fallout (mg/L)
Cl_s = average chloride concentration in soil pore water (mg/L)
P = average annual precipitation (mm/yr).

The product of Cl_p and P is termed the chloride deposition rate. The estimation of recharge using Equation (B.1) is commonly referred to as the chloride mass balance (CMB) method. Key assumptions are that chloride is nonreactive (i.e., no sorption or transformation), subsurface flow is piston-type flow, and precipitation and chloride deposition rates are steady during the time interval of interest.

Cl_p can be measured directly through long-term collection of precipitation and dry fallout or inferred by measurements of pre-bomb $^{36}Cl/Cl$ ratios in the soil profile (i.e., ratios below the peak $^{36}Cl/Cl$ associated with atmospheric nuclear weapons testing in the 1950s). Cl_s is commonly measured directly by extracting chloride from the sediment sample using a combination of contact with deionized water and filtration of the resulting supernatant fluid.

B.1.2 Chlorine-36

Between 1952 and 1958, nuclear weapons testing added a significant quantity of ^{36}Cl to the atmosphere. The subsequent fallout of this weapons-derived ^{36}Cl produced a noticeable peak in the ratio of ^{36}Cl to chloride in soil pore water relative to ratios in pore water prior to and after the cessation of testing. The peak in the $^{36}Cl/Cl$ ratio in soil pore water is also known as the bomb pulse.

Because ^{36}Cl behaves as a conservative tracer, the $^{36}Cl/Cl$ peak in the soil pore water can be used as a time marker relative to the peak atmospheric concentration of ^{36}Cl and thus used to estimate pore water flux. Pyrch (1998) estimated that the center of mass of elevated ^{36}Cl deposition between 30 and 50 degrees north latitude (the Hanford Site sits at 46.5 degrees north latitude) occurred in the year 1957. Atmospheric deposition during this time was about 1,000 times the natural rate of production of ^{36}Cl. With an estimate of time of peak ^{36}Cl deposition and the assumption that transport is driven strictly by advective flow, the recharge rate can be estimated by dividing the amount of water in the soil profile above the depth of peak pore water $^{36}Cl/Cl$ ratio by the elapsed time from the peak ^{36}Cl fallout year (1957) to the soil sampling date.

B.1.3 ^{18}O and ^{2}H

The isotopes ^{18}O and ^{2}H are commonly used for recharge analysis because they both are stable and occur naturally in measurable quantities. As soil pore water evaporates, the isotopic composition of oxygen and hydrogen changes due to the preferential loss of lighter isotopes during evaporation. This results in the pore water having a larger ratio of heavier oxygen and hydrogen isotopes compared to the composition of precipitation. The magnitude of the isotopic shift is related to the level of pore water evaporative loss. With information about the isotopic departure and the appropriate model, a recharge estimate may be attained.

The isotopic compositions of oxygen and hydrogen are commonly reported relative to a standard material, using

$$\delta(‰) = \left(\frac{R_{sample} - R_{std}}{R_{std}} \right) \times 1000 \tag{B.2}$$

where R is the isotopic ratio (R = $^{18}O/^{16}O$ or $^{2}H/^{1}H$) of the sample and standard material. Vienna-Standard Mean Ocean Water (V-SMOW) has been used as the standard material for past Hanford-related ^{18}O and ^{2}H recharge studies (e.g., DePaolo et al. 2004).

The ^{18}O and ^{2}H isotopes are ideal for estimating recharge when evaporation is the only process removing water from the soil. In situations where plants are present, however, the method is more cumbersome. Plants extract water from the soil without discriminating between isotopes (Dawson et al. 2002). To account for this extraction requires additional assumptions regarding plant water uptake behavior, and the additional assumptions increase the uncertainty of the recharge estimate. When plants are present but not considered in the analysis of recharge, the recharge estimates derived are considered upper boundary values.

B.1.4 Strontium

A relatively new environmental tracer method for recharge estimation involves the use of the strontium isotopic ratio ($^{87}Sr/^{86}Sr$) in pore water. From the ground surface to the groundwater, the strontium isotopic ratio of the pore water increasingly reflects the composition of the bulk sediment. The rate of change of the pore water strontium isotopic ratio toward the isotopic ratio of the bulk sediment, per specific depth interval, provides a measure of the pore water strontium flux relative to the sediment strontium dissolution flux. This rate of change can be used to estimate the ratio of the weathering rate to the fluid flux, from which the recharge rate can be estimated. As developed, the method provides estimates for rates that existed hundreds to thousands of years ago.

B.1.5 Uranium

Another relatively new environmental tracer method for recharge estimation is to use the ratio of two uranium isotopes ($^{234}U/^{238}U$) in pore water. Like the strontium isotopic ratio, the uranium isotopic ratio in soil pore water is affected by the pore water flux and the dissolution rate of the bulk sediment. Unlike the strontium method, α-recoil loss of ^{234}U from the bulk sediment also contributes to the U isotope composition of the soil pore water. The α-recoil loss of ^{234}U is associated with the energetic α-decay of ^{238}U. With information on the uranium isotope ratio profile, bulk weathering rate, and the rate of ^{234}U addition to the pore water by α-recoil, an estimate of recharge can be made.

B.2 Published Tracer-Based Recharge Estimates

Ten studies have been published documenting the use of the above-mentioned environmental tracers to estimate recharge beneath the Central Plateau at the Hanford Site (Table B.1). In some of the studies, recharge was estimated also using other recharge analysis methods (e.g., lysimetry and computer simulation) to provide estimates for conditions for which tracer data were unavailable or inappropriate. Figure B.1 shows the locations of the published tracer-based recharge estimates. A synopsis of the published studies is presented here.

Table B.1. Tracer Recharge Studies Relevant to the Hanford Central Plateau

Title	Authors	Source	Publish Year
Geochemical Estimates of Paleorecharge in the Pasco Basin: Evaluation of the Chloride Mass Balance Technique	Murphy et al.	*Water Resources Research* 32(9):2853–2868	1996
Using Chloride and Chlorine-36 as Soil-Water Tracers to Estimate Deep Percolation at Selected Locations on the U.S. Department of Energy Hanford Site, Washington	Prych	*United States Geological Survey* Water Supply Paper 2481	1998
Recharge Data Package for the Immobilized Low-Activity Waste 2001 Performance Assessment	Fayer et al.	PNNL-13033	1999
Vadose Zone Infiltration Rate at Hanford, Washington Inferred from Sr Isotope Measurements	Maher et al.	*Water Resources Research* 39(8):3	2003
Evaporation Effects on Oxygen and Hydrogen Isotopes in Deep Vadose Zone Pore Fluids at Hanford, Washington	DePaolo et al.	*Vadose Zone Journal* 3:220–232	2004
Recharge Data Package for the 2005 Integrated Disposal Facility Performance Assessment	Fayer and Szecsody	PNNL-14744	2004
Chloride Mass Balance: Cautions in Predicting Increased Recharge Rates	Gee et al.	*Vadose Zone Journal* 4:72–78	2005
Dissolution Rates and Vadose Zone Drainage from Strontium Isotope Measurements of Groundwater in the Pasco Basin, WA Unconfined Aquifer	Singleton et al.	*Journal of Hydrology* 321:39–58	2006
U-Sr Isotopic Speedometer: Fluid Flow and Chemical Weathering Rates in Aquifers	Maher et al.	*Geochimica et Cosmochimica Acta* 70:4417–4435	2006
Assessment of the Chloride Mass Balance Technique: Accounting for Gravel Content	Keller et al.	PNNL-16346	2007

B.2.1 Murphy et al. (1996)

Murphy et al. (1996) used the CMB method to estimate recharge near the Yakima Barricade in the 200 West Area and the Wye Barricade approximately 8 km south of the 200 East Area. In addition, using deep $^{36}Cl/Cl$ ratios representing ratios prior to the anthropogenic fallout of the 1950s, the authors estimated a chloride deposition rate [$Cl_p \times P$ in Equation (B.1)] used in estimating recharge with the CMB method.

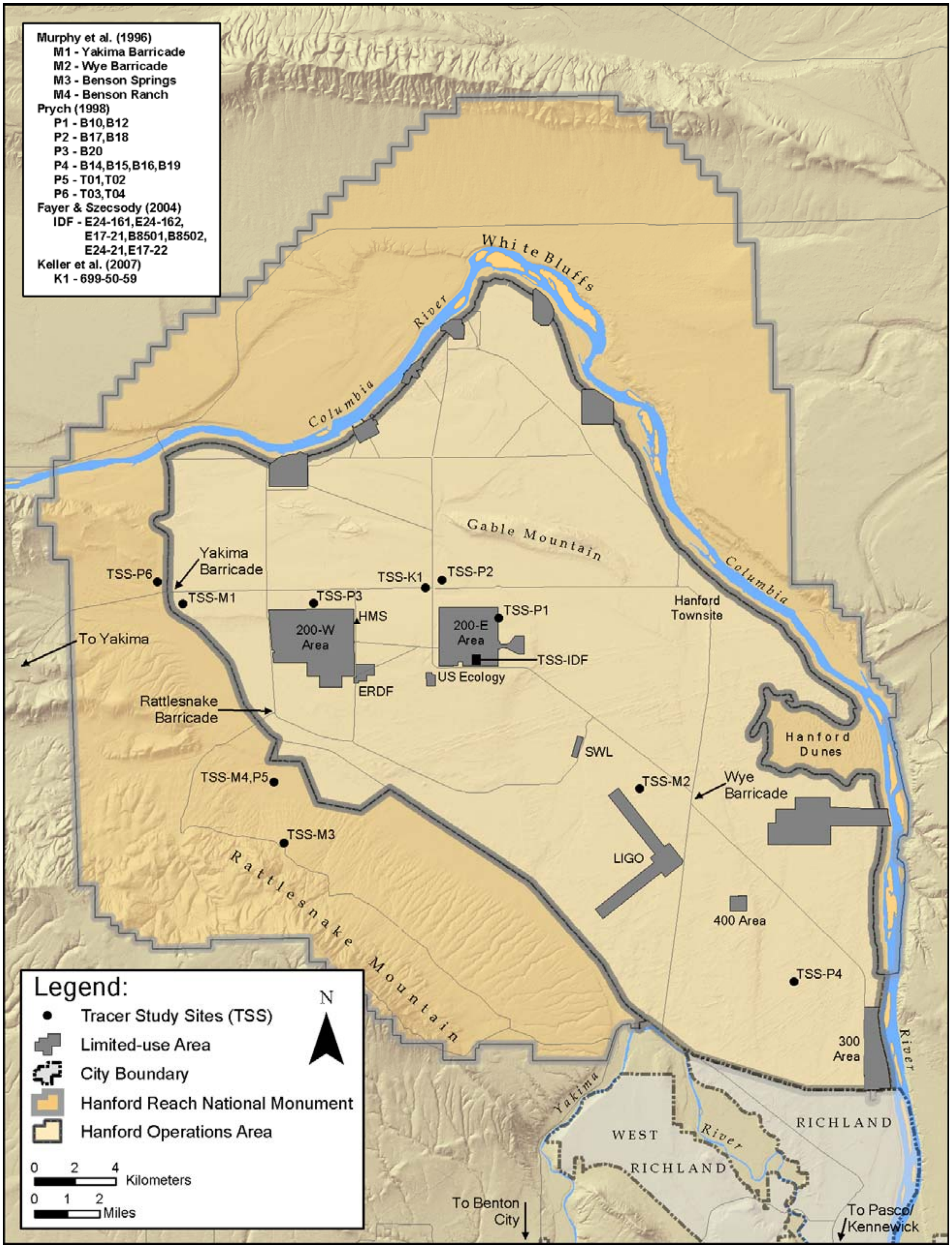

Figure B.1. Locations of Data Sources on the Central Plateau for Tracer-Based Recharge Estimates

Three boreholes were drilled between 1990 and 1994 at the Yakima Barricade site in an area dominated by mature sagebrush and overlain by a layer of silt loam less than 1 m thick. At this location, the ^{36}Cl/Cl data from five sampling depths were used to estimate an average atmospheric chloride deposition rate of 40.0 ± 7.3 mg/m^2/yr. Using this deposition rate and the pore water chloride profile from the three Yakima Barricade boreholes, Murphy et al. (1996) estimated recharge rates of 0.011, 0.012, and 0.013 mm/yr for the three boreholes.

A single borehole was drilled in 1994 at the Wye Barricade site in an area characterized by Murphy et al. (1996) as a stabilized sand dune with a coarse sand surface soil and vegetation consisting predominantly of cheatgrass and Sandberg's blurgass, with a sparse cover of gray rabbitbrush and sagebrush. Using the deposition rate estimated from the Yakima Barricade borehole and the measured pore water chloride profile at this location, Murphy et al. (1996) estimated a recharge rate of 4.0 mm/yr.

B.2.2 Pyrch (1998)

Prych (1998) used the CMB method to estimate recharge rates for 13 boreholes around the Hanford Site. For four of those boreholes, Pyrch (1998) also estimated recharge using the chlorine-36 (bomb pulse) method. Five of the Prych boreholes were located on or near the Central Plateau and all had loamy sand or sandy loam soils. Four boreholes were in silt loam soil on the lower slopes and base of the Rattlesnake Hills. The remaining four boreholes were in a layered loamy sand above sand soil at a location called the Grass Site.

Two of the Central Plateau boreholes (B10 and B12) were drilled in 1990 near the Liquid Effluent Recovery Facility (LERF), adjacent to the eastern boundary of the 200 East Area. Vegetation at this location was primarily sagebrush and sparse grass. The surface soil was identified by Pyrch (1998) as either a Burbank loamy sand or Ephrata sandy loam to a depth of approximately 0.6 m. Underlying this soil is approximately 60 m of Hanford formation sand and gravels. Two boreholes (B17 and B18) were drilled in 1990 directly north of the 200 East Area, with surface soil and vegetation described by Pyrch (1998) as being similar to those for the boreholes near the LERF. Underlying the soil is approximately 50 m of coarse Hanford formation sediments, with significant gravel content existing intermittently. The fifth Central Plateau borehole (B20) was drilled between 1991 and 1992 just outside the northern boundary of the 200 West Area. Prych (1998) described the surface soil and vegetation at this borehole as being similar to that near the LERF. Underlying the soil are sedimentary deposits extending below the water table. Boreholes T01 and T02 were drilled in 1990 and 1991 in locations dominated by sagebrush vegetation and silt loam soil. The boreholes are on the lower slopes of the Rattlesnake Hills. Boreholes T03 and T04 were drilled in 1990 and 1991 on terraced land at the base of the Rattlesnake Hills. Both boreholes were in silt loam soil between hummocks that defined the sampling location. Vegetation between the hummocks was primarily sparse grass, with sagebrush vegetation on the top of the hummocks. The Grass Site boreholes (B14, B15, B16, and B19) were drilled in 1990 and 1991 in a soil consisting of approximately 0.6 m of loamy sand above approximately 9 m of sand. Grasses were the dominant vegetation at this site, which Prych (1998) described as having a notable absence of shrubs or other deep-rooted plants, possibly because of a wildfire.

Using deep ^{36}Cl/Cl ratios from borehole B20 and two additional boreholes near the base of Rattlesnake Mountain, Pyrch (1998) estimated the chloride deposition rate using the same analysis as Murphy et al. (1996). The estimated deposition rates ranged from 33 to 39 mg/m^2/yr and averaged 35 mg/m^2/yr. The estimated deposition rate using only borehole B20 was 33 mg/m^2/yr.

Pyrch (1998) used the deposition rate derived from the B20 borehole to estimate recharge for the five 200 Area boreholes. The recharge estimates ranged from 0.012 mm/yr (at B20) to 0.3 mm/yr (at B18) and averaged 0.087 mm/yr. These estimates are based on the peak chloride concentrations, all of which occurred in the depth range from 3 to 10 m. Below the peak pore water chloride concentrations used for these recharge estimates, the pore water chloride concentrations decreased. Prych (1998) surmised that the lower pore water chloride concentrations deeper in the vadose zone were representative of previous surface conditions (e.g., soil texture and vegetation) that produced greater historic recharge rates. Based on pore water chloride concentrations located below the depth of the peak concentration, the estimated recharge rates ranged from 5.5 mm/yr (at B12) to 0.66 mm/yr (at B20), with the arithmetic average recharge for all five boreholes being 2.4 mm/yr.

Prych (1998) estimated recharge for borehole B20 using the chlorine-36 method. The center of mass of the pore water ^{36}Cl in this borehole was at a depth of 1.50 m, which translates into a recharge estimate of 2.6 mm/yr. Prych (1998) considered this estimate to be an upper limit for recharge, given the ^{36}Cl center of mass was still located in the root zone where the continued removal of soil water through plant transpiration would reduce recharge.

Recharge estimates from the boreholes in silt loam soil ranged from 0.008 mm/yr (at T02) to 0.11 mm/yr (at T04). The average recharge estimate from all four silt loam boreholes was 0.0388 mm/yr. For borehole T04, the pore water chloride concentrations that were below the depth of the peak concentration yielded an estimated recharge rate of 0.42 mm/yr. For boreholes T02 and T03, the chlorine-36 was still within the root zone. Using the center of mass of pore water ^{36}Cl, the upper limit recharge values for boreholes T02 and T03 were estimated to be 3.4 and 2.1 mm/yr, respectively.

Recharge estimates from the Grass Site boreholes ranged from 0.39 mm/yr (at B14) to 2.0 mm/yr (at B15). The average estimated recharge from all four Grass Site boreholes was 1.2 mm/yr. Using the center of mass of pore water ^{36}Cl profile from borehole B19, a recharge of 5.1 mm/yr is estimated. Prych (1998) cautioned that the actual recharge rate may be larger because the entire anthropogenic ^{36}Cl profile was not captured in this borehole.

B.2.3 Fayer et al. (1999)

In the appendix of Fayer et al. (1999), a study is reported in which the CMB method was used to estimate recharge at the IDF Site (formerly called the immobilized low-activity waste, or ILAW site) in the 200 East Area. Two boreholes were drilled in 1995 to a depth of approximately 18 m at a location consisting of 0.3 to 0.6 m of sand overlying Hanford formation material composed of gravelly sands and sands. Both boreholes were located in a dense stand of mature sagebrush. Four additional boreholes were drilled in the sand dune that makes up the south boundary of the IDF site. Vegetation was mature sagebrush but sparser than at the location of the first two boreholes.

In addition to the boreholes, a trench was dug in 1995 at the IDF site and soil samples were collected from the backhoe at specified depths. The $^{36}Cl/Cl$ profile from this trench was used to establish an atmospheric chloride deposition rate of 38.4 mg/m^2/yr using the same analysis as Murphy et al. (1996). This deposition rate was used by Fayer et al. (1999) and Fayer and Szecsody (2004) in their CMB analysis on cores collected at the IDF site.

The possibility of increased surface deposition of chloride from emissions from an upwind coal-fired power plant raised concerns about the usefulness of the pore water chloride data for recharge estimation using the CMB method. To avoid concerns about possible anthropogenic chloride deposition, Fayer et al. (1999) restricted their analysis to pore water chloride at depths below the chloride peak concentration. Their assumption was that the deeper chloride was deposited prior to operation of the coal-fired power plant beginning in 1945. CMB analysis of pore water chloride data from the two 1995 boreholes in the sagebrush vegetated area produced recharge estimates of 0.16 and 0.71 mm/yr. These estimates are lower than the range of values (0.66 to 5.5 mm/yr) predicted by Pyrch (1998) for similar surface conditions elsewhere on the Central Plateau but consistent with his estimates, given the small recharge being estimated and errors associated with the CMB method. For the three boreholes in the dune area, CMB analysis produced recharge estimates of 1.0, 1.1, and 2.1 mm/yr. The recharge estimates in the dune area are greater than estimates for the mature sagebrush locations as would be expected, given the reduced vegetation at the dune site.

The ^{36}Cl/Cl profile from the 1995 trench data was used also to obtain an estimate of recharge for Rupert sand surface soils. The ^{36}Cl/Cl center of mass was identified at approximately 1.5 m, still within the root zone where evapotranspiration processes can affect downward water flux. Because the center of mass was still within the root zone, a recharge rate was not estimated. However, an upper limit of recharge of 5 mm/yr was established.

B.2.4 Maher et al. (2003)

Maher et al. (2003) estimated recharge at a disturbed location near the S-SX Tank Farm in the 200 West Area using the ^{87}Sr/^{86}Sr profile obtained from samples collected from borehole 299-W22-48. The vadose zone at this location is composed of sand-dominated and gravel-dominated Hanford formation sediments to a depth of approximately 40 m. Below that are Cold Creek unit sediments and Ringold Formation sediments. Along with the ^{87}Sr/^{86}Sr ratios, Maher et al. (2003) used estimates of sediment weathering rates and dissolution fluxes as input to a strontium transport model in which the pore water flux was optimized. The estimated isotopic dissolution flux produced an estimated recharge rate of 7 ± 3 mm/yr.

B.2.5 DePaolo et al. (2004)

From the same borehole utilized by Maher et al. (2003), DePaolo et al. (2004) estimated recharge using ^{18}O pore water composition. Isotopic analysis of the soil pore water showed an average δ^{18}O shift of +3.6 ± 0.8‰ relative to that of winter precipitation. Using this information and a “semiquantitative” model developed for non-vegetated sites by DePaolo et al. (2004), a maximum recharge rate for this area was estimated to range between 35 to 60 mm/yr. This estimate represents an average recharge for the past 100 to 1,000 years whereas the estimate by Maher et al. (2003) using ^{87}Sr/^{86}Sr data represents a past record of 1,000 to 10,000 years. DePaolo et al. (2004) suggested that the difference between the two estimates may be due to different prevailing surface conditions during the different analysis periods.

B.2.6 Fayer and Szecsody (2004)

Fayer and Szecsody (2004) report CMB results from two boreholes drilled at the IDF site in mature sagebrush stands. The surface around one borehole (C3177) had experienced disturbance decades prior to drilling and, as a result, had only marginal plant cover. The surface at the site of the second borehole (C3826) also was disturbed and had not had an established shrub population since approximately 1987. The authors noted, however, the presence at both sites of deep-rooted annual plants like tumble mustard and tumbleweed. Borehole C3177 was sited in a Burbank loamy sand surface soil while borehole C3826 was placed in Rupert sand surface soil. As with other recharge estimates made with CMB at the IDF site (Fayer et al. 1999), concerns exist about increased chloride deposition due to operation of a coal-fired power plant directly adjacent to the IDF site. To address this concern, the recharge analysis was restricted to the pore water chloride concentrations below the depth of the peak concentration. Recharge from these two boreholes was estimated to be 0.24 mm/yr from the C3177 profile and 0.62 mm/yr from the C3826 profile.

Using ^{18}O data acquired from cores taken from the C3826 borehole and the empirical relationship between recharge and oxygen-18 developed by Barnes and Allison (1988) and Barnes et al. (1989), Fayer and Szecsody (2004) estimated a recharge rate of 1.3 mm/yr. As expected, this estimate is larger than the recharge rate of 0.62 mm/yr predicted by the CMB method. The isotope-derived estimate is expected to be larger because this method does not account for uptake of the isotope by surface vegetation.

B.2.7 Gee et al. (2005)

Gee et al. (2005) evaluated the CMB method for estimating recharge under high recharge conditions by comparing long-term average drainage from a lysimeter to estimates derived using the CMB method. The 300 North Lysimeter is located about 22 km (14 mi) southeast of the 200 East Area. The lysimeter is filled with Hanford formation sand and has been maintained free of vegetation since 1978. Gee et al. (2005) reported that, over 26 years, the average drainage was 62 mm/yr. They found that the pore water chloride concentrations of lysimeter sediment samples yielded recharge estimates that were lower than the measured drainage by factors ranging from two to eight. The authors evaluated multiple causes, including anion exclusion, sample handling and analysis, and mineral dissolution, but were unable to explain the discrepancy.

B.2.8 Singleton et al. (2006)

Singleton et al. (2006) used ^{87}Sr/^{86}Sr groundwater results from seven Hanford groundwater monitoring wells, along with weathering rate estimates and an advective dispersive transport model for strontium (Johnson and DePaolo 1997), to estimate recharge for a transect that runs in a northwest to southeast direction across the Hanford Site (but south of the Central Plateau). The authors estimated that recharge ranges from 0 to 1.4 mm/yr along the transect from northwest to southeast. At the point closest to the 200 Areas (specifically the southern end of the 200 West Area), the recharge estimate ranged from 0 to 0.4 mm/yr. As pointed out by Maher et al. (2003), these rates apply for thousands of years ago.

B.2.9 Maher et al. (2006)

Using data reported by Maher et al. (2003) and DePaolo et al. (2004) for borehole 299-W22-48, Maher et al. (2006) estimated recharge using ratios of ^{234}U/^{238}U along with estimates of bulk sediment dissolution rates and α-recoil loss of ^{234}U. Additionally, the authors used the strontium isotope ratio data from Maher et al. (2003) to constrain the reaction and infiltration model. Application of this information produced a recharge estimate of 5 ± 2 mm/yr, which they stated was applicable for past periods from hundreds to thousands of years ago.

B.2.10 Keller et al. (2007)

Keller et al. (2007) estimated recharge using pore water chloride measurements from nearly continuous cores collected from well 699-50-59 just outside the northwest corner of the 200 East Area. The location of the well is dominated by mature sagebrush and a variety of annual and perennial grasses. Core samples were collected to a depth of 15 m (49.2 ft). The samples consisted of gravel-dominated sediments ranging from 58% to 85% gravel content. Subsamples of the cores were processed for anion analysis using a 0.7 to 1 water to sediment extract ratio. The extract subsamples captured the entire sample particle size composition (i.e., both >2- and <2-mm constituents). Using a chloride deposition rate of 40 mg/m^2/yr (Murphy et al. 1996) and the average pore water chloride concentration of 16.3 mg/L, Keller et al. estimated a recharge rate of 2.5 mm/yr.

B.3 Published Borehole Tracer Data for Recharge Estimation

Published borehole data packages were examined to locate pore water chloride data that could support estimates of recharge using the CMB method. To be clear, none of the boreholes was drilled and sediment samples collected and characterized with recharge determination as a primary objective, so we had no expectations about the data sets being complete and robust for such analyses. However, the data sets existed and offered the potential for estimating recharge rates with little additional effort.

This section describes the methods used and the recharge rates estimated from the data packages. The section finishes by summarizing the results with respect to sites that have had only surface disturbance and those which have been deeply disturbed (e.g., tank farms). Both metric and English units of measurement are reported in this section to be consistent with the referenced borehole data packages in which English units were used.

B.3.1 Methods

This section describes the approach used to select data packages to support recharge estimation, the borehole sampling and chloride analysis methods used for the data packages, and the method for determining the chloride deposition rate to be used in the estimation of recharge.

B.3.1.1 Borehole Selection

Published data packages for boreholes in and around SST WMAs were examined to locate pore water chloride data that could support estimates of recharge using the CMB method. None of the boreholes was drilled for recharge estimation purposes, so we recognize that the data sets may not be ideal. However, the lack of any other recharge-related data for the SST environments argues that even imperfect chloride data sets ought to be examined to extract whatever value possible and guide any future recharge estimation endeavors.

The appropriateness of the chloride data sets in terms of their usefulness for CMB analyses was determined based on the depth resolution of data (i.e., number of samples per interval of interest), the method of sample collection (core sample versus grab sample), and the presence of waste contaminants within the depth interval of interest. Chloride data sets that contained sufficient data resolution to capture the pore water chloride profile and that appeared to be unaffected by contamination from anthropogenic activities were considered for further evaluation. Table B.2 lists the criteria used to discriminate among the data packages.

Table B.2. Rating Criteria for Evaluating Borehole Data Packages for Recharge Estimation Using the CMB Method

Rating	Rating Criteria
Ideal	• Continuous core samples collected in liners from the soil surface to a minimum of 15 m • Strong evidence that the profile from the surface to 15 m has not come into contact with waste liquids
Adequate	• Core samples collected in liners; alternatively, grab samples for which in situ water contents are known • At least five samples for the interval between the bottom of the evapotranspiration zone and a depth of 15 m • No suspicions (e.g., records of leaks; pore water chemistry) that the sediment between the soil surface and 15 m has been contacted by waste liquids
Marginal	• Core samples collected in liners; alternatively, grab samples for which in situ water contents are known • At least two samples for the interval between the bottom of the evapotranspiration zone and a depth of 15 m • Possibility exists that part or all of the sediment from the soil surface to 15 m was in contact with waste liquids, but pore water chemistry suggests contaminants are no longer present in the interval of interest
Inadequate	• Isolated grab samples with no measure of in situ water content • No more than one 1 sample for the interval between the bottom of the evapotranspiration zone and a depth of 15 m • Contaminants present in interval of interest

A summary of the reports analyzed and the ratings assigned is presented in Tables B.3 through B.5. Of the 30 characterization reports reviewed, none was considered ideal. Two reports were considered adequate, and eight were considered marginal. As noted above, given the lack of other recharge-related data, even the marginal chloride data sets were analyzed for this report. Of the two reports rated as adequate, the chloride data in the Horton et al. (2003) characterization report for borehole C3177 had been used previously to estimate recharge at the IDF site by Fayer and Szecsody (2004) and was discussed in Section B.2.

The following sections refer to two types of borehole identification numbers that sometimes create confusion, so the following explanation is provided. Monitoring wells at Hanford are identified by three sets of characters separated by hyphens (e.g., 299-W22-50) that identify the Hanford area in which the well exists. The character set 299 identifies the well as being in the 200 Area. The characters E and W at the start of the second character set identify the well as being in either the 200 East Area (E) or the 200 West Area (W). The remaining numbers in the second character set identify the survey sheet containing the well (e.g., sheet 22 in the example above). The third set of characters indicates that the borehole is a dry well if the set is a number from 51 to 299, otherwise the borehole is a ground water monitoring well. In addition to the Hanford well numbering system, each well is assigned a start card number by the Washington Department of Ecology. This number is an alphanumeric sequence such as C3830. In general, if the well was drilled solely for the purpose of collecting sediment samples and then decommissioned, it is referenced by a start card number only and does not possess a Hanford well number.

Table B.3. Hanford Borehole Characterization Studies Rated Adequate for CMB Estimation of Recharge

Report Number	Title	Authors	Publication Date
PNNL-13757-1	*Characterization of Vadose Zone Sediment: Uncontaminated RCRA Borehole Core Samples and Composite Samples*	Serne et al.	February 2002
PNNL-14289	*Geochemistry of Samples from Borehole C3177 (299-E24-21)*	Horton et al.	May 2003

Table B.4. Hanford Borehole Characterization Studies Rated Marginal for CMB Estimation of Recharge

Report Number	Title	Authors	Publication Date
PNNL-13757-2	*Characterization of Vadose Zone Sediment: Borehole 299-W23-19 [SX-115] in the S-SX Waste Management Area*	Serne et al.	February 2002
PNNL-13757-3	*Characterization of Vadose Zone Sediment: Borehole 41-09-39 in the S-SX Waste Management Area*	Serne et al.	February 2002
PNNL-14083	*Characterization of Vadose Zone Sediment: Borehole 299-E33-45 Near BX-102 in the B-BX-BY Waste Management Area*	Serne et al.	December 2002
PNNL-14121	*Characterization of Vadose Zone Sediment: RCRA Borehole 299-E33-338 Located Near the B-BX-BY Waste Management Area*	Lindenmeier et al.	June 2003
PNNL-14594	*Characterization of Vadose Zone Sediments Below the TX Tank Farm: Probe Holes C3830, C3831, C3832, and 299-W10-27*	Serne et al.	April 2004
PNNL-14849	*Characterization of Vadose Zone Sediments Below the T Tank Farm: Probe Holes C4104, C4105, 299-W10-196, and RCRA Borehole 299-W11-39*	Serne et al.	September 2004
PNNL-15503	*Characterization of Vadose Zone Sediments Below the C Tank Farm: Borehole C4297 and RCRA Borehole 299-E27-22*	Brown et al.	September 2006

B.3.1.2 Sampling

Figures B.2 and B.3 show the locations of the boreholes in the 200 West and East Areas, respectively, that were drilled to provide samples for SST evaluations. Vadose zone sediment samples were collected using a split-spoon sampler either 0.76 m (2.5 ft) or 0.38 m (1.25 ft) long. A dry drilling technique was used so that the in situ water content could be preserved. The 0.76-m (2.5-ft) split-spoon sampler contained two Lexan liners 10.1 cm (4 in.) in diameter and 0.30 m (1 ft) long. The 0.38-m (1.25-ft) split-spoon sampler contained one Lexan liner 6.4 cm (2.5 in.) in diameter and 0.30 m (1 ft) long. If the sediment was expected to be contaminated, stainless steel liners were used instead of Lexan. Sampling depth and resolution varied widely between boreholes. In most instances, cores were not collected shallower than 4 m (13.1 ft), although a few boreholes did have cores from shallower depths. If continuous coring was being performed, a set of two cores 0.30 m (1 ft) long was collected every 0.76 m (2.5 ft). Typically only one core of the pair was analyzed, so the resolution was 0.76 m (2.5 ft). For all boreholes, continuous coring was not performed throughout the entire length of the borehole. Instead, samples were collected either periodically or continuously for specific segments of the borehole.

Table B.5. Hanford Borehole Characterization Studies Rated Unsuitable for CMB Estimation of Recharge

Report Number	Title	Authors	Publication Date
PNNL-11515	*Borehole Data Package for Well 699-37-47A, PUREX Plant Cribs, CY 1996*	Lindberg et al.	February 1997
PNNL-12127	*Borehole Data Package for 216-U-12 Crib Well 299-W22-79*	Horton and Williams	March 1999
PNNL-13198	*Borehole Data Package for the 216-S-10 Pond and Ditch Well 299-W26-13*	Horton et al.	May 2000
PNNL-13199	*Borehole Data Package for Wells 299-E33-334 and 299-E33-335 at Single-Shell Tank Waste Management Area B-BX-BY*	Horton	May 2000
PNNL-13200	*Borehole Data Package for Well 299-W22-48, 299-W22-49, and 299-W22-50 at Single-Shell Tank Waste Management Area S-SX*	Horton and Johnson	May 2000
PNNL-13589	*Borehole Data Package for Calendar Year 2000-2001 RCRA Wells at Single-Shell Tank Waste Management Area S-SX*	Horton and Johnson	August 2001
PNNL-13590	*Borehole Data Package for Calendar Year 2000-2001 RCRA Wells at Single-Shell Tank Waste Management Area T*	Horton and Hodges	August 2001
PNNL-13591	*Borehole Data Package for Calendar Year 2000-2001 RCRA Wells at Single-Shell Tank Waste Management Area TX-TY*	Horton and Hodges	September 2001
PNNL-13757-4	*Characterization of Vadose Zone Sediment: Slant Borehole SX-108 in the S-SX Waste Management Area*	Serne et al.	February 2002
PNNL-13826	*Borehole Data Package for Calendar Year 2001 RCRA Wells at Single-Shell Tank Waste Management Area TX-TY*	Horton	March 2002
PNNL-13827	*Borehole Data Package for Calendar Year 2001 RCRA Wells at Single-Shell Tank Waste Management Area B-BX-BY*	Horton	March 2002
PNNL-13828	*Borehole Data Package for Calendar Year 2001 RCRA Wells at Single-Shell Tank Waste Management Area U*	Horton	March 2002
PNNL-13829	*Borehole Data Package for Calendar Year 2001 RCRA Wells at Single-Shell Tank Waste Management Area S-SX*	Horton	April 2002
PNNL-13830	*Borehole Data Package for Calendar Year 2001 RCRA Wells at Single-Shell Tank Waste Management Area T*	Horton	March 2002
PNNL-14119	*Characterization of Vadose Zone Sediment: Borehole 299-E33-46 Near Tank B-110 in the B-BX-BY Waste Management Area*	Serne et al.	December 2002
PNNL-14128	*Characterization of Vadose Zone Sediment: Borehole C3103 Located in the 216-B-7A Crib and Selected Samples from Borehole C3104 Located in the 216-B-38 Trench Near the BX Tank Farm*	Lindenmeier et al.	December 2002
PNNL-14249	*Data Package for Calendar year 2002 RCRA Groundwater Monitoring Wells at Single-Shell Tank Waste Management Area TX-TY*	Horton	April 2003
PNNL-14255	*Data Package for Groundwater Monitoring Well 299-W15-43 at the 200-ZP-1 Operable Unit*	Horton	April 2003
PNNL-14656	*Borehole Data Package for Four CY 2003 RCRA Wells 299-E27-4, 299-E27-21, 299-E27-22, and 299-E27-23 at Single-Shell Tank, Waste Management Area C, Hanford Site, Washington.*	Williams and Narbutovskih	June 2004
PNNL-15141	*Investigation of Accelerated Casing Corrosion in Two Wells at Waste Management Area A-AX*	Brown et al.	August 2005
PNNL-15617	*Characterization of Vadose Zone Sediments from C Waste Management Area: Investigation of the C-152 Transfer Line Leak*	Brown et al.	January 2007

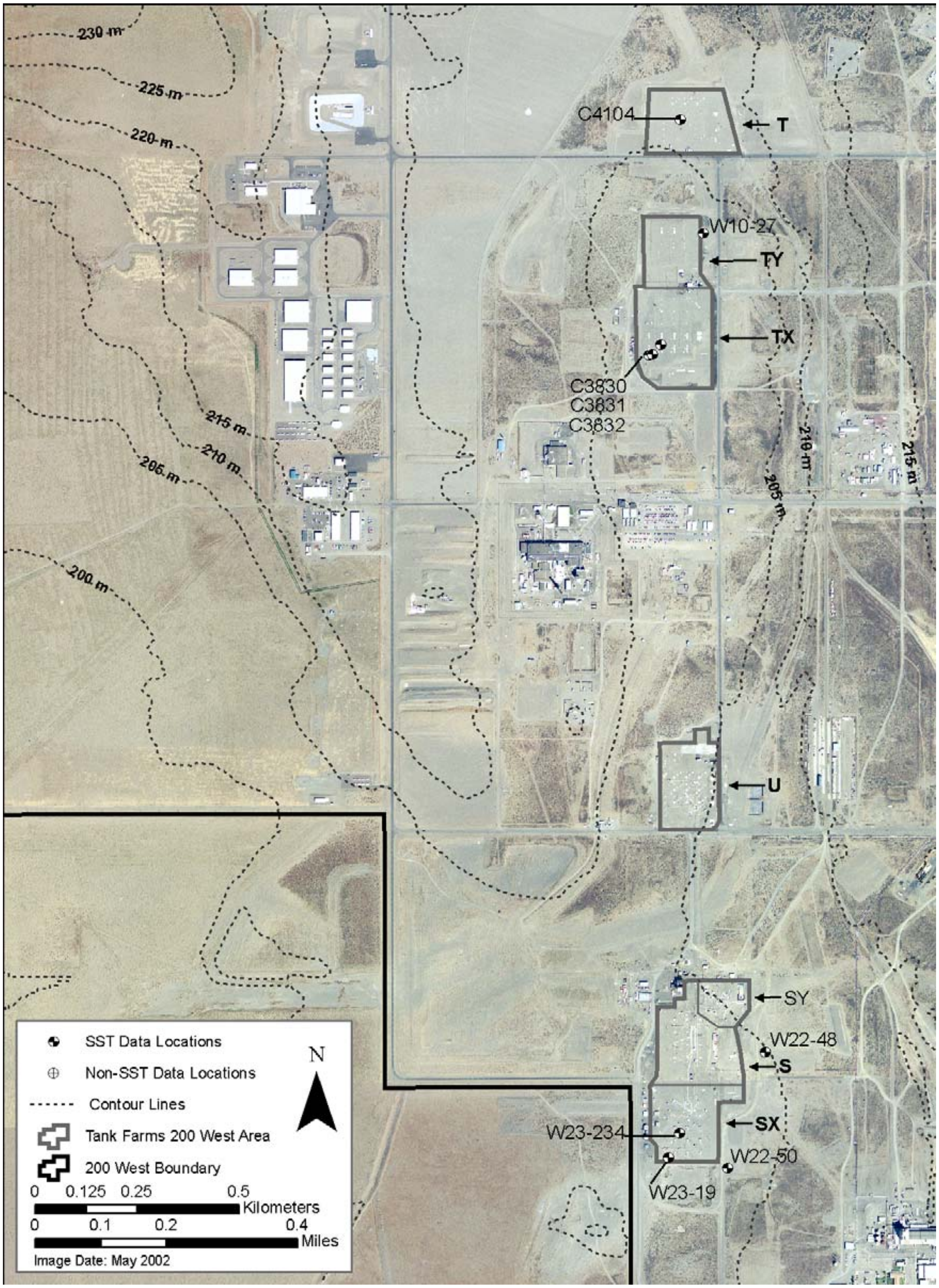

Figure B.2. Locations of SST Boreholes in the 200 West Area That Provided Data for Recharge Estimation

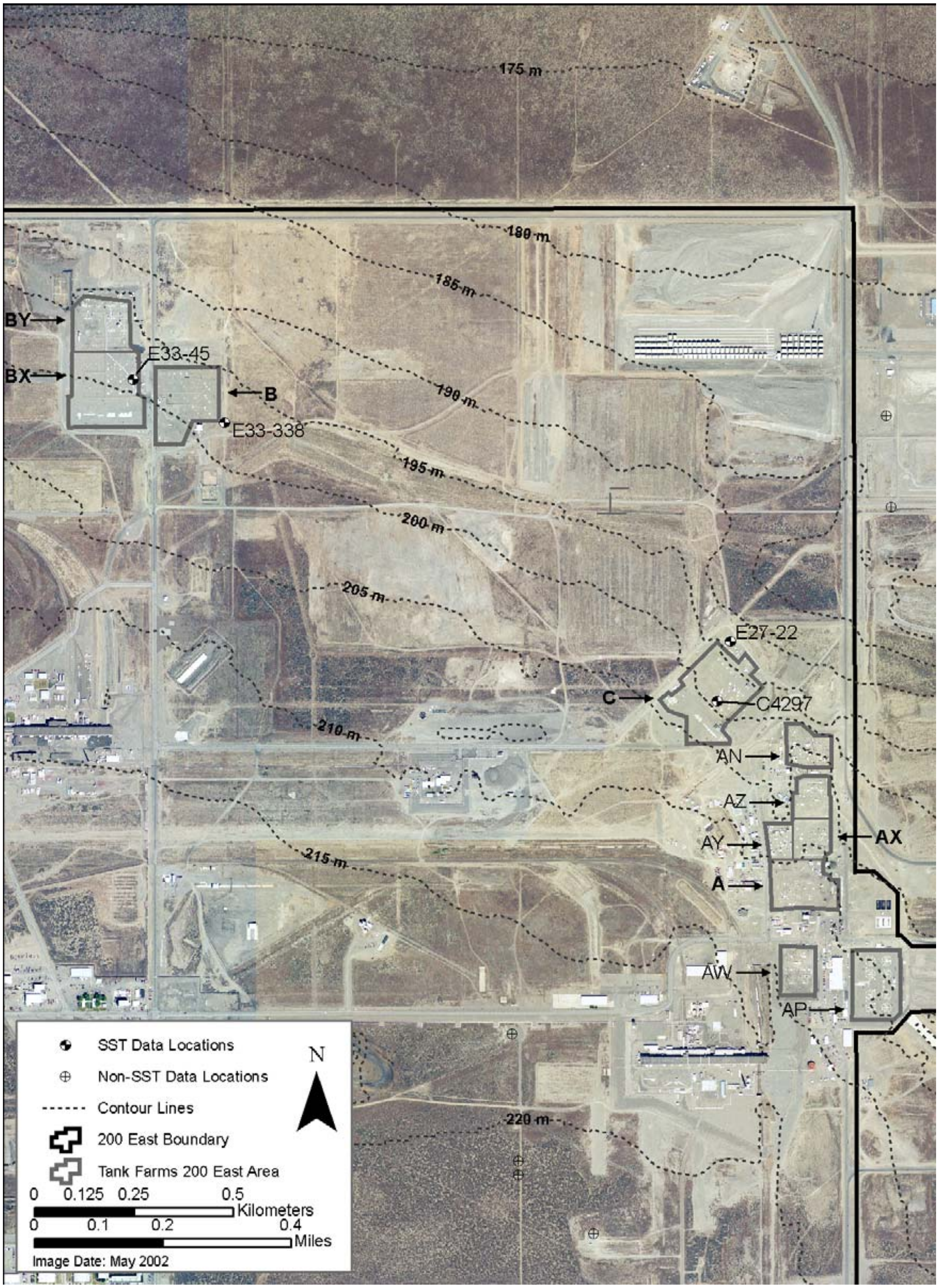

Figure B.3. Locations of SST Boreholes in the 200 East Area That Provided Data for Recharge Estimation

Once cores were recovered from the borehole, three steps were taken to preserve the integrity of the cores and minimize moisture loss. First, open space at the end of the cores was filled with aluminum foil to keep the sediment from moving within the liner during transport and handling. Second, the core ends were secured with plastic end caps that were sealed to the liners with duct tape. Finally, the sealed cores were stored in coolers on site to maintain constant temperature and minimize any moisture loss till the cores could be transported to the laboratory for further analyses.

B.3.1.3 Chloride Analysis

In the laboratory, the sediment was accessed by cutting and opening the core liners lengthwise or by pushing the sediment out of the liner. The sediment was then subsampled for anion and water content analyses. The anion subsample received an addition of deioinized water equivalent to the oven-dry weight of that subsample, which was estimated using the oven-dry weight of the separate subsample. The ratio of water added to sediment is known as the extraction ratio; for these boreholes, that ratio was 1:1.

The deionized water was added to the sediment in a screw cap jar, immediately after which the jar was sealed, shaken briefly by hand, and then placed on a mechanical orbital shaker for one hour. The samples were allowed to settle until the supernatant liquid had cleared. The supernatant liquid was decanted and filtered through a 0.45-μm membrane. The filtered leachate, called the extract, was analyzed for anions using an ion chromatograph with a quantification limit of 0.02 mg/L. The chloride concentration of the extract, Cl_{ext}, was converted into a pore water chloride concentration, Cl_s, as follows:

$$Cl_s = \frac{Cl_{ext}}{w} D \tag{B.3}$$

where w is the gravimetric water content (g/g) and D is the sediment-to-water extract ratio. For more specific details on sample preparation and analysis, the reader is directed to the individual borehole data package reports.

As noted in the discussion of Prych (1998), the gravel fraction of a sample can contain significant quantities of chloride, directly adhering to either the gravel or to a coating of fine particles attached to the gravel. For the borehole data packages, no attempt was made to separate out the gravel content. However, the shaker bottles used in the extraction procedure have a throat opening that is approximately 16 mm in diameter. This means that only those gravel particles in the bulk sample that were greater than 16 mm in diameter were eliminated from the extraction subsample. Reported chloride concentrations in the data packages have not been corrected for the gravel fraction greater than 16 mm. In all instances, either the borehole data packages do not provide particle size information for the samples of interest or the published particle size information does not allow for the determination of the fraction of particles that are greater than 16 mm in diameter. As a general statement, based on the limited grain-size information provided in the data packages and knowledge of the typical Hanford formation sediments, most of the samples used in the CMB analysis to be described in this section contained only small amounts of particles greater than 16 mm, so little to no gravels were excluded from the analysis.

B.3.1.4 Chloride Deposition Rate

Previous studies have determined different chloride deposition rates for Hanford. Murphy et al. (1996) estimated a deposition rate of 40 ± 7 mg/m^2/yr. Prych (1998) reported four estimates (33, 34, 39, 39 mg/m^2/yr) for four locations around Hanford. Prych used 33 mg/m^2/yr to calculate recharge for the five boreholes in the 200 Areas. Fayer et al. (1999) collected samples from a newly opened pit at the

Integrated Disposal Facility (IDF) site. Based on eleven $^{36}Cl/Cl$ ratio measurements in the depth range from 2.4 to 4.6 m, Fayer et al. (1999) estimated a chloride deposition rate of 38.4 mg/m^2/yr and used it to estimate recharge for five boreholes at the IDF site. Fayer and Szecsody (2004) used the same rate to estimate recharge for two more boreholes at the IDF site. Keller et al. (2007) used 40 mg/m^2/yr (the estimate determined by Murphy et al. 1996).

For this RFI recharge data package, the rate of 38.4 mg/m^2/yr was used to estimate recharge using the chloride data in the borehole data packages. The difference between this rate and the previous studies is less than 20%, so no attempt was made to update the previously published recharge estimates using a common chloride deposition rate.

B.3.2 Surface Disturbed Sediment

Of the borehole data packages with a quality rating of 2 (marginal) or better, four of the reports contain pore water chloride information obtained from boreholes in areas where the soil is assumed not to have been excavated. However, given the boreholes' proximity to certain tank farms, the surfaces of many of these boreholes likely were disturbed as a result of tank farm construction activities or site operation activities. Common disturbances include removal of vegetation and the compactive and disruptive effects of vehicle traffic. The following summarizes the five boreholes sited in surface disturbed areas that have been identified as having a chloride data set potentially useful for estimating recharge.

B.3.2.1 PNNL-13757-1

Serne et al. (2002a) provided chloride data for two boreholes, 299-W22-48 and 299-W22-50, near WMA S-SX. Samples from borehole 299-W22-48 were used by DePaolo et al. (2004), Maher et al. (2003), and Maher et al. (2006) to estimate recharge using isotopic analyses. This borehole is approximately 64 m east of the S Tank Farm fence (Figure B.4). The second borehole, 299-W22-50, is in an area approximately 25 m east of the SX Tank Farm fence and approximately 290 m south of borehole 299-W22-48 (Figure B.5). Figures B.4 and B.5 show that both boreholes are in areas that have no vegetation and show extensive evidence of surface modification with gravelly material. Borehole 299-W22-48 was completed in December 1999, and 299-W22-50 was completed in January 2000.

Table B.6 lists the extract concentration—Cl_{ext} (mg/L) and field moisture content, w (mg/mg)—for both boreholes (Serne et al. 2002a). The pore water chloride concentrations, Cl_s (mg/L), were calculated using Equation (B.3).

Figure B.6 shows that the pore water chloride concentration profile for borehole 299-W22-48 exhibits elevated chloride concentrations near the surface, indicative of concentrated pore water chloride due to evapotranspiration conditions. Borehole 299-W22-48 has a relatively uniform chloride profile between 1.83 m (6 ft) and 14.33 m (47 ft), with chloride concentrations ranging from 3.02 mg/L to 25.63 mg/L. Below 14.33 m (47 ft), the chloride concentration profile increases to a peak of 117.27 mg/L and is somewhat erratic. The chloride profile between 1.83 m (6 ft) to 14.33 m (47 ft) is assumed to have developed under current climatic and surface conditions. The average pore water chloride concentration in this depth range is 9.89 mg/L. Using that average pore water concentration, Equation (B.1) yields an estimated recharge rate of 3.88 mm/yr.

Figure B.7 shows that the pore water chloride concentration profile for borehole 299-W22-50 exhibits a relatively uniform chloride profile between 6.1 m (20 ft) to 15.54 m (51 ft), with chloride concentrations

Figure B.4. Surface Conditions Around Borehole 299-W22-48 (April 19, 2007)

Figure B.5. Surface Conditions Around Borehole 299-W22-50 (April 19, 2007)

Table B.6. Soil Water Extract Chloride, In Situ Moisture Content, and Calculated Pore Water Chloride for Boreholes 299-W22-48 and 299-W22-50

Borehole 299-W22-48					Borehole 299-W22-50				
Depth (m)	Depth (ft)	Extract Chloride (mg/L)	Moisture Content (wt%)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Extract Chloride (mg/L)	Moisture Content (wt%)	Pore Water Chloride (mg/L)
0.30	1.0	1.27	3.87	32.82	6.10	20.0	0.45	7.33	6.14
1.83	6.0	0.42	9.12	4.61	6.86	22.5	1.08	14.46	7.47
2.90	9.5	0.39	4.86	8.02	7.62	25.0	0.76	5.80	13.10
3.81	12.5	0.32	6.21	5.15	8.38	27.5	1.88	11.47	16.39
4.42	14.5	0.30	7.63	3.93	9.14	30.0	4.85	12.56	38.61
5.18	17.0	0.35	5.92	5.91	9.91	32.5	1.62	8.72	18.58
5.94	19.5	0.72	7.70	9.35	10.67	35.0	0.82	8.94	9.17
6.71	22.0	0.49	5.03	9.74	11.43	37.5	0.67	6.86	9.77
7.47	24.5	0.37	2.22	16.67	12.19	40.0	0.57	5.07	11.24
8.23	27.0	0.49	6.04	8.11	12.95	42.5	0.99	7.14	13.87
8.99	29.5	0.32	10.60	3.02	13.72	45.0	0.82	7.87	10.42
9.75	32.0	0.26	5.33	4.88	14.48	47.5	0.83	10.60	7.83
10.45	34.3	0.39	3.85	10.13	15.54	51.0	0.45	7.62	5.91
11.28	37.0	1.43	5.58	25.63	16.00	52.5	0.85	2.55	33.33
12.04	39.5	0.65	7.90	8.23	16.61	54.5	1.34	2.08	64.42
12.80	42.0	0.43	2.80	15.36	17.07	56.0	0.54	4.25	12.71
13.56	44.5	0.42	2.53	16.60	18.29	60.0	9.33	3.02	308.94
14.33	47.0	0.24	1.88	12.77	20.57	67.5	4.84	7.51	64.45
15.24	50.0	0.84	5.25	16.00	23.16	76.0	4.40	10.30	42.72
16.31	53.5	0.53	2.41	21.99	29.26	96.0	23.70	8.97	264.21
17.07	56.0	1.05	3.91	26.85	33.83	111.0	11.47	13.86	82.76
17.53	57.5	1.72	2.83	60.78	35.05	115.0	14.41	10.17	141.69
22.71	74.5	5.06	6.56	77.13	39.62	130.0	2.01	6.31	31.85
27.89	91.5	4.83	19.14	25.24	41.15	135.0	2.31	10.00	23.10
30.94	101.5	5.48	21.62	25.35					
32.46	106.5	6.45	5.50	117.27					
35.20	115.5	2.28	5.29	43.10					
41.45	136.0	1.32	5.70	23.16					

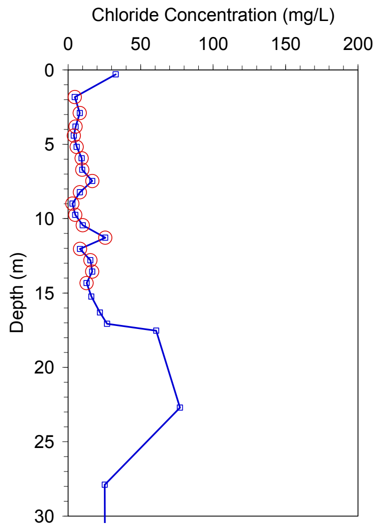

Figure B.6. Pore Water Chloride Concentration Profile for Borehole 299-W22-48. Circles identify sample depths used in estimating recharge.

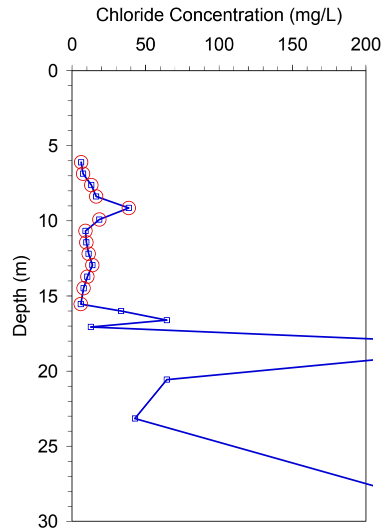

Figure B.7. Pore Water Chloride Concentration Profile for Borehole 299-W22-50. Circles identify sample depths used in estimating recharge.

ranging from 5.91 mg/L to 38.61 mg/L. Below 15.54 m (51 ft), the chloride concentration profile varies considerably to the final sample depth; the peak concentration is 308.94 mg/L at 18.29 m (60 ft). The average pore water chloride concentration between 6.1 and 15.54 m (20 to 51 ft) was calculated to be 12.96 mg/L. Using that average pore water chloride concentration, Equation (B.1) yields an estimated recharge rate of 2.96 mm/yr.

The pore water chemistry from both boreholes shows no indication that the sediments from these boreholes have been contacted by contaminants. Any variations in pore water chloride concentrations are therefore more easily attributable to variations in chloride deposition, recharge rates, and material properties.

The chloride-derived estimated recharge rate of 3.91 mm/yr for borehole 299-W22-48 is consistent with the estimates of 7 ± 3 mm/yr by Maher et al. (2003) and 5 ± 2 mm/yr by Maher et al. (2006) using strontium and uranium isotope analyses on samples from the same borehole. In contrast, the estimated recharge rate is considerably less than the 35 to 60 mm/yr recharge estimated by DePaolo et al. (2004) using isotopic analysis of oxygen from 299-W22-48 samples. Lysimeter measurements of recharge under tank-farm–like conditions (gravel-covered surfaces) suggest rates ought to be 80 to 100 mm/yr (Fayer and Szecsody 2004). The discrepancy among these estimates arises from at least three factors: period of effective measurement, chloride deposition rate, and material property differences.

The strontium and uranium isotope methods apply to recharge conditions that existed hundreds to thousands of years ago (Maher et al. 2006). The methods are not suited to estimating recharge conditions in the past 60 years, i.e., during the period of Hanford operations and remediation.

The chloride deposition rate was determined using deep ^{36}Cl data that reflects conditions thousands of years ago. Modern chloride deposition rates have not been determined. Fayer and Szecsody (2004) suggested that local facilities (e.g., coal-fired power plants) could have raised the local chloride deposition rate.

The lysimeter data are specific to the materials in the lysimeters. Those materials are similar to surface materials in the tank farms, but subtle differences may exist that could alter the results. Smoot et al. (1989) identified some particle size data from tank farm surfaces that suggest the materials are slightly finer textured than expected. If so, the effect would be to increase evaporation and reduce drainage (and thus recharge) relative to the coarser materials in the lysimeters.

The matric potential data from both boreholes (W22-48 and W22-50) were high. In the top 30 m of the boreholes, matric potential values ranged from −0.01 to −0.4 MPa, with an overall average of −0.08. Such high values are indicative of draining conditions relative to the water table location at 65 m.

B.3.2.2 PNNL-14121

Lindenmeier et al. (2003) provided chloride data for sediment cores from borehole 299-E33-338, which was installed in July and August 2001 just outside the southeastern corner of WMA B. Figure B.8 shows the location of the borehole relative to the tank farm fence. It also shows the site lacks vegetation and the surface has been modified with gravelly material.

Table B.7 lists the pore water chloride concentration for borehole 299-E33-338 as a function of depth. Figure B.9 shows the pore water chloride concentration profiles for 299-E33-338. Pore water analysis results do not suggest that the sampled sediment has come into contact with tank waste liquid. The chloride concentration peaks at 223.42 mg/L at the 15.71-m (51.6-ft) depth, coinciding with the contact between the H1 and H2 Hanford formations. The two chloride concentration measurements above the chloride peak were used to determine the average pore water chloride concentration of 21.27 mg/L. Using that average pore water concentration, Equation (B.1) yielded an estimated recharge rate of 1.81 mm/yr.

Figure B.8. Surface Conditions Around Borehole 299-E33-338 (April 19, 2007)

Table B.7. Pore Water Chloride for Borehole 299-E33-338

Borehole 299-E33-338		
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
4.88	16.0	32.09
5.49	18.0	10.45
15.71	51.6	223.42
23.71	77.8	21.07
25.54	83.8	9.62
27.81	91.3	15.62
32.86	107.8	1.80
35.33	115.9	131.42
40.69	133.5	15.10
49.01	160.8	57.12

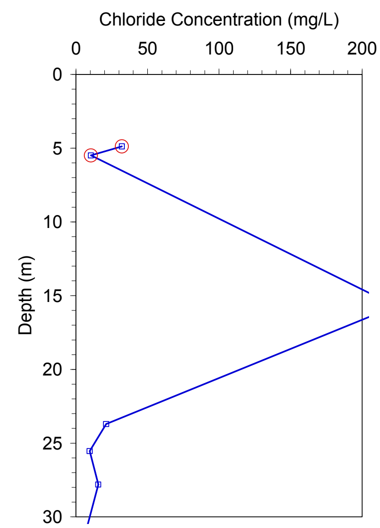

Figure B.9. Pore Water Chloride Concentration Profile for Borehole 299-E33-338. Circles identify sample depths used in estimating recharge.

The matric potential data from borehole E33-338 were high. In the top 30 m of the boreholes, matric potential values ranged from −0.002 to −0.14 MPa, with an overall average of −0.02. Such high values are indicative of draining conditions relative to the water table location at 78 m.

B.3.2.3 PNNL-14594

Serne et al. (2004a) provided some chloride data from borehole 299-W10-27 located about 10 m (30 ft) outside the east fence of WMA TY. The borehole was installed between January and March 2001. Figure B.10 shows that the surface at this borehole is unvegetated and modified with a gravelly material. Below the gravel, the sediments consist of a thin layer of eolian material, Hanford formation, Cold Creek unit, and Ringold Formation. Reported geophysical monitoring and pore water analytical results suggest that the sediments at this location have not been exposed to tank waste liquids.

Table B.8 gives the pore water chloride concentration for borehole 299-W10-27 calculated from reported extract concentrations and field moisture content. Figure B.11 shows that the pore water chloride concentration profile for 299-W10-27 has a peak concentration of 48.68 mg/L at 26.91 m (88.3 ft). The location of this peak is just above the interface between Hanford formation and Cold Creek unit sediments. The average chloride concentration in the zone above this peak, i.e., 16.25 to 18.59 m (53.3 to 61.0 ft), is 9.70 mg/L. Using this average pore water chloride concentration, Equation (B.1) yielded an estimated recharge rate of 3.96 mm/yr.

Figure B.10. Surface Conditions Around Borehole 299-W10-27 (April 19, 2007). The TY Tank Farm is behind the fence.

Table B.8. Soil Water Extract Chloride, Field Moisture Content, and Calculated Pore Water Chloride for Borehole 299-W10-27

Borehole 299-W10-27				
Depth (m)	Depth (ft)	Chloride (µg/g dry soil)	Moisture Content (wt%)	Pore Water Chloride (mg/L)
16.25	53.3	0.64	9.61	6.66
16.31	53.5	0.90	9.61	9.37
18.59	61.0	0.90	6.88	13.08
26.91	88.3	1.66	3.41	48.68
28.19	92.5	0.48	8.66	5.54
30.48	100.0	0.37	7.25	5.10
34.59	113.5	0.61	3.08	19.64
35.81	117.5	0.22	6.05	3.64

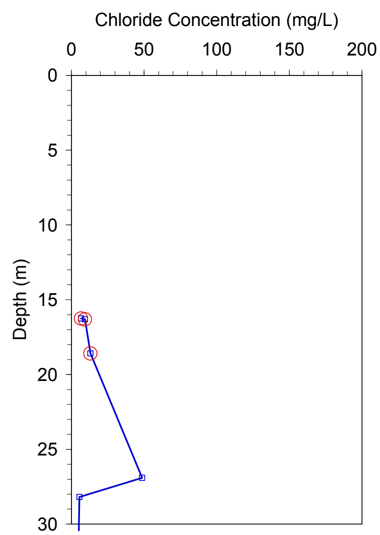

Figure B.11. Pore Water Chloride Concentration Profile for Borehole 299-W10-27. Circles identify sample depths used in estimating recharge.

The matric potential data from borehole W10-27 were high. In the top 30 m of the boreholes, matric potential values ranged from −0.01 to −0.4 MPa, with an overall average of −0.15. Such high values are indicative of draining conditions relative to the water table location at 69 m.

B.3.2.4 PNNL-15503

Brown et al. (2006) provided pore water chloride information for borehole 299-E27-22, which was drilled about 23 m (75 ft) north of WMA C in August 2003. Figure B.12 shows that the site is unvegetated and that the surface was modified with a gravelly material. Beneath the graveled surface, the sediment is about 1.2 m (4 ft) of eolian sand on top of Hanford formation and undifferentiated Cold Creek unit/Ringold Formation sediments. Geophysical logging of the borehole detected no radioactive contaminants.

Table B.9 lists the pore water chloride concentration for borehole 299-E27-22 calculated from reported extract concentrations and field moisture content. Although no documented releases have occurred at the location of this borehole, the authors state that elevated electrical conductivity and elevated anion concentrations in the extract suggest that the sediment has come into contact with non-radiological waste. Figure B.13 shows that the pore water chloride concentration profile for borehole 299-E27-22 is relatively uniform between 8.53 and 13.87 m (28 and 45.5 ft). Below that depth range, the chloride concentration rises to 217.90 mg/L at the 14.63-m (48.0-ft) depth (which corresponds to the location of a fine-textured layer) and continues to be high down to the base of the borehole. Assuming that sediment above 14.63 m was no longer affected by past leak events, the pore water chloride concentrations in that interval between 8.53 to 13.87 m (28 to 45.5 ft) averaged 23.56 mg/L. Using that average, Equation (B.1) yielded an estimated recharge rate of 1.63 mm/yr.

Figure B.12. Surface Conditions Around Borehole 299-E27-22 (April 19, 2007)

Table B.9. Soil Water Extract Chloride, Field Moisture Content, and Calculated Pore Water Chloride for Borehole 299-E27-22

Borehole 299-E27-22				
Depth (m)	Depth (ft)	Chloride (µg/g sediment)	Moisture Content (wt%)	Pore Water Chloride (mg/L)
8.53	28.0	0.48	2.68	17.91
12.34	40.5	1.01	4.15	24.34
13.87	45.5	0.95	3.34	28.44
14.63	48.0	27.15	12.46	217.90
15.39	50.5	4.42	2.39	184.94
23.77	78.0	3.22	2.04	157.84
24.99	82.0	20.70	10.72	193.10
26.06	85.5	6.06	1.79	338.55
29.11	95.5	3.67	2.52	145.63
30.63	100.5	2.61	1.48	176.35
42.52	139.5	6.62	2.10	315.24
44.35	145.5	4.12	2.42	170.25
48.92	160.5	3.16	2.20	143.64

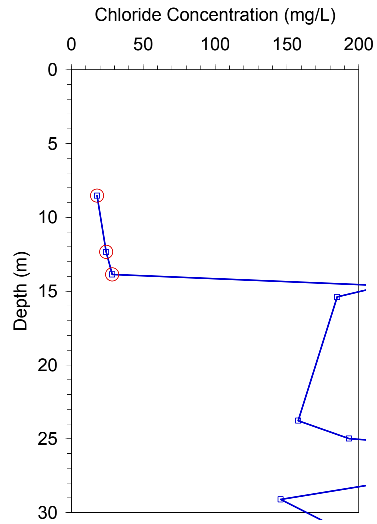

Figure B.13. Pore Water Chloride Concentration Profile for Borehole 299-E27-22. Circles identify sample depths used in estimating recharge.

The matric potential data from borehole W10-27 were generally high. In the top 30 m of the boreholes, matric potential values ranged from 0.0 to −3.6 MPa, with an overall average of −0.34 and a median value of −0.15. Excluding the few low potential values, the preponderance of high values is indicative of draining conditions relative to the water table location at 70 m.

B.3.3 Deeply Disturbed Sediment

Eight boreholes with a quality rating of 2 were located within previously excavated sediment of tank farms, providing information that can be used to assess recharge for deeply disturbed sediment. The backfill material was stockpiled from the removal of soil to create a pit 12 to 19 m (39.4 to 62.3 ft) for placement of the tanks. As the tanks were completed, this stockpiled material was placed around the tanks and brought to grade. As a result of this disturbance and general homogenization of sediment, the pore water composition was likely altered from natural steady-state conditions. Long-term drainage data for similar coarse-textured soils (Fayer and Szecsody 2004; Gee et al. 2005; Ward et al. 2005) support the assumption that drainage through the backfill material is sufficiently large that given the time between backfill placement and sample collection (i.e., 50 to 60 years), the backfill pore water should represent modern natural recharge conditions.

For all boreholes identified here, the sediment at some depth was exposed to tank waste fluid from a tank leak, a transfer line leak, or a tank overfill event. In all cases, only the chloride data within the backfill was considered for recharge analysis. In some of those cases, the leak occurred on the soil surface or within the backfill. In those instances, it was assumed that recharge through the coarse, nonvegetated tank farm surfaces was large enough for an amount of time sufficient to have flushed the tank leak liquids to a depth in the soil profile that would allow use of the shallower chloride data.

B.3.3.1 PNNL-13757-2 and PNNL-13757-3

Serne et al. (2002a, 2002b) reported data for two boreholes inside WMA S-SX, 299-W23-19 and 299-W23-234. Borehole 299-W23-19 is in the southwestern corner of the WMA S-SX, approximately 3 m (10 ft) southwest of SST SX-115, which was confirmed to have leaked in 1965. Borehole 299-W23-19 was drilled and split spoon samples collected in August 1999. At this borehole site, the backfill material was observed to exist from the surface to a depth of 18.7 m (61.4 ft).

Borehole 299-W23-234 is located between SSTs SX-108, -109, -111, -112. In 1965, SX-109 was determined to have leaked. Borehole 299-W23-234 was originally installed in December 1996 and deepened between September and December 1997. Core samples identify backfill material at this location to 15.9 m (52.1 ft).

Table B.10 lists the pore water chloride concentrations for boreholes 299-E23-19 and 299-W23-234. Pore water chloride concentration profiles for boreholes 299-E22-19 and 299-W23-234 are presented in Figures B.14 and B.15, respectively. Analysis of soil pore water chemistry from borehole 299-W23-19

Table B.10. Soil Pore Water Chloride for Boreholes 299-W23-19 and 299-W23-234

Borehole 299-W23-19			Borehole 299-W23-234		
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
10.04	33.0	1.60	7.77	25.5	3.7[(a)]
16.61	54.5	12.20	13.56	44.5	10.70
17.51	57.5	14.40	17.22	56.5	5.22
18.82	61.8	7.00	18.75	61.5	6.34
19.28	63.3	4.30	19.96	65.5	10.90
19.43	63.8	4.70	20.12	66.0	4.89
19.89	65.3	5.60	21.18	69.5	8.80
20.56	67.5	5.20	22.71	74.5	10.50
21.17	69.5	6.70	24.23	79.5	31.30
21.47	70.5	7.90	25.15	82.5	358.00
21.63	71.0	8.40	27.43	90.0	2992.00
22.33	73.3	56.80	29.11	95.5	4074.00
22.48	73.8	50.40	31.24	102.5	1903.00
22.63	74.3	47.40	33.07	108.5	2990.00
22.94	75.3	44.20	34.14	112.0	4212.00
24.16	79.3	34.45	38.83	127.4	613.00
24.31	79.8	26.70			
24.46	80.3	27.80			
26.21	86.0	16.40			
26.85	88.1	39.50			
28.96	95.0	44.50			
30.19	99.1	43.75			
30.50	100.1	79.10			
31.38	103.0	43.40			
31.99	105.0	30.40			
32.63	107.1	70.80			
(a) Sample below detection limit. Used detection limit of instrument.					

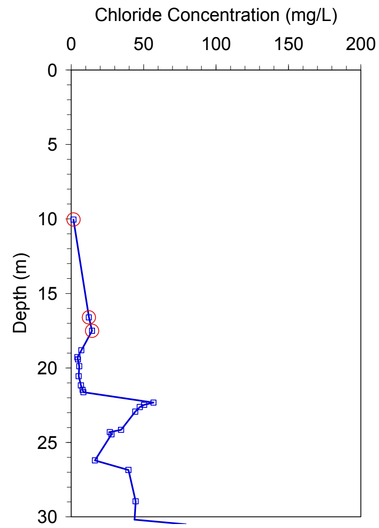

Figure B.14. Pore Water Chloride Concentration Profile for Borehole 299-W23-19. Circles identify sample depths used in estimating recharge.

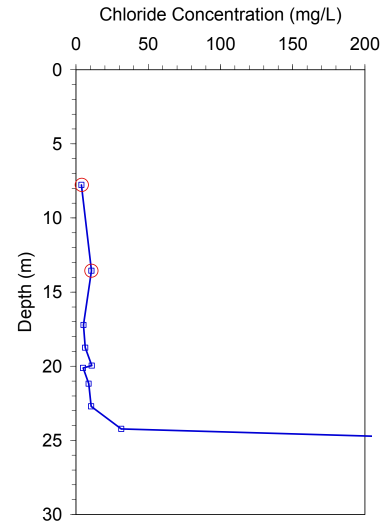

Figure B.15. Pore Water Chloride Concentration Profile for Borehole 299-W23-234. Circles identify sample depths used in estimating recharge.

verified that tank leak fluid had contacted sediment between approximately 19.8 and 47.5 m (65.0 and 155.8 ft) deep. Given this information, only chloride data above 18.0 m (59.1 ft) were used to estimate recharge. This constraint limited the analysis to three samples, which covered the depth interval from 10.04 to 17.51 m (33 to 57.5 ft). Using the average chloride concentration of 9.4 mg/L in this depth interval, Equation (B.1) yielded an estimated recharge rate of 4.09 mm/yr.

The pore water chemistry of samples from borehole 299-W23-234 suggests that sediments were exposed to tank leak fluid from the base of the tanks at approximately 16.8 m (55.1 ft) to a depth exceeding 47.5 m (155.8 ft). Therefore, only the two chloride data values above 16.8 m (55.1 ft) were used to estimate recharge. One of those values was below the quantification limit; for this report, the concentration was set at the quantification limit (in this case, 3.7 mg/L). Using the average chloride concentration of 7.20 mg/L above 16.8 m, Equation (B.1) yielded an estimated recharge rate of 5.33 mm/yr.

The matric potential data from borehole W23-19 were high. In the top 30 m of the boreholes, matric potential values ranged from 0.0 to −0.14 MPa, with an overall average of −0.04. Such high values are indicative of draining conditions relative to the water table location at 65 m.

B.3.3.2 PNNL-14083

Serne et al. (2002d) provided chloride data from borehole 299-E33-45, located in WMA BX. This borehole was installed approximately 21.3 m (70 ft) northeast of SST BX-102 between November 2000 and January 2001. In 1951, SST BX-102 experienced an overfill event that resulted in tank waste liquid entering the surrounding sediment. The borehole is on the eastern edge of the tank farm excavation, resulting in the backfill being only 3 m (9.8 ft) thick. Beneath the backfill material are Hanford formation sediments.

Table B.11 lists the pore water chloride concentrations as a function of depth for borehole 299-E33-45. Figure B.16 shows that the pore water chloride concentration profile is relatively uniform between 3.24 to 19.20 m (10.6 to 63 ft). Assuming that meteoric water has flushed the tank waste from the soil profile past the depth of 19.20 m (63 ft) and that flow was vertical across the backfill–Hanford formation interface, the pore water chloride concentration from 3.24 to 19.20 m (10.6 to 63 ft) was used to evaluate recharge at this location. Using the average chloride concentration of 17.17 mg/L in this depth interval, Equation (B.1) yielded an estimated recharge rate of 2.24 mm/yr.

The matric potential data from borehole E33-45 were high. In the top 30 m of the boreholes, matric potential values ranged from −0.001 to −0.31 MPa, with an overall average of −0.02. Such high values are indicative of draining conditions relative to the water table location at 78 m.

Table B.11. Pore Water Chloride for Borehole 299-E33-45

Borehole 299-E33-45		
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
3.24	10.6	24
6.35	20.8	12
9.70	31.8	18
12.81	42.0	21
15.74	51.6	15
19.20	63.0	13
21.71	71.2	53
22.37	73.4	59
23.06	75.7	157
23.83	78.2	91
24.18	79.3	62
27.02	88.7	35
30.51	100.1	130
33.88	111.1	19
36.28	119.0	22
36.62	120.1	33

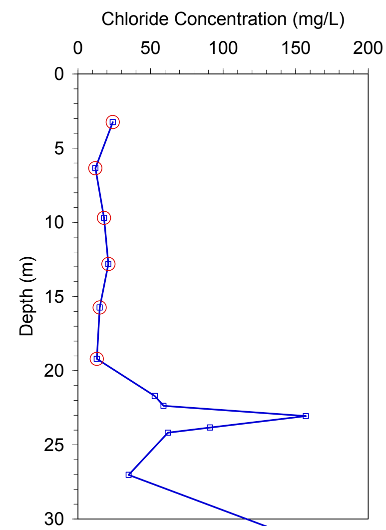

Figure B.16. Pore Water Chloride Concentration Profile for Borehole 299-E33-45. Circles identify sample depths used in estimating recharge.

B.3.3.3 PNNL-14594

Pore water chloride data are presented in Serne et al. (2004a) for three boreholes placed in WMA TX to evaluate possible leaks from SSTs TX-105, -107, and -104. Borehole C3830 is approximately 12.2 m (40.0 ft) southwest of SSST TX-105; C3831 is approximately 12 m (39.4 ft) southwest of SST TX-107; and C3832 is approximately 6 m (19.7 ft) southeast of SST TX-104. Borehole C3830 was drilled between August and September 2002. Borehole C3831 was drilled between July and August 2002, while C3832 was drilled in May 2002. Backfill sediment at each location was identified to a depth of 15.8 (52.0 ft).

Table B.12 gives the pore water chloride concentrations for boreholes C3830, C3831, and C3832. Pore water chloride concentration profiles for boreholes C3830, C3831, and C3832 are presented in Figures B.17, B.18, and B.19, respectively. Pore water chemistry of cores from all three boreholes support the assumption that sediment near the base of the tanks and below has come into contact with tank waste liquid. As a result, the recharge analysis was limited to only those samples collected above the bottom of the tanks. For C3830, the depth range was 4.87 to 12.83 m (16.0 to 42.1 ft) and the resulting average pore water concentration was 7.09 mg/L. For C3831, the depth range was 4.87 to 8.81 m (16.0 to 28.9 ft) and the resulting average pore water chloride concentration was 18.83 mg/L. For C3832, the depth interval was 5.26 to 13.77 m (17.3 to 45.2 ft) and the resulting average pore water chloride concentration was 7.40 mg/L. Using those average chloride concentrations, Equation (B.1) yielded estimated recharge rates of 5.42, 2.04, and 5.19 mm/yr for boreholes C3830, C3831, and C3832, respectively.

Table B.12. Pore Water Chloride for Boreholes C3830, C3831, and C3832

Borehole								
C3830			C3831			C3832		
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
4.87	16.0	6.41	4.87	16.0	18.09	5.26	17.3	5.10
8.81	28.9	9.06	6.68	21.9	20.92	8.88	29.1	13.44
12.83	42.1	5.80	8.81	28.9	17.49	11.32	37.1	4.51
14.38	47.2	12.04	14.04	46.1	308.09	13.77	45.2	6.53
14.82	48.6	31.77	15.86	52.0	5.05	15.85	52.0	40.74
16.47	54.0	8.19	16.21	53.2	31.43	16.49	54.1	8.09
			18.27	59.9	149.85	18.64	61.2	25.34
17.69	58.0	23.45	18.64	61.2	225.15	19.19	63.0	14.96
18.09	59.4	14.08	20.58	67.5	65.24	21.33	70.0	3.33
20.46	67.1	115.37				23.27	76.3	69.75
22.14	72.6	111.91	20.79	68.2	27.72			
23.78	78.0	296.38	21.21	69.6	30.36	23.48	77.0	31.71
24.96	81.9	215.65	22.86	75.0	41.51	24.33	79.8	22.93
26.55	87.1	104.08	23.67	77.7	56.74	25.61	84.0	26.93
30.49	100.0	222.97	23.99	78.7	38.54	26.84	88.1	13.91
30.90	101.4	159.64	26.21	86.0	112.68	28.67	94.1	13.07
			27.25	89.4	49.08	29.47	96.7	25.83
31.72	104.1	134.49	28.65	94.0	85.58	32.01	105.0	78.57
			29.89	98.1	43.10	33.61	110.3	85.03
			30.87	101.3	66.45	33.84	111.0	90.10
			31.29	102.7	100.57	35.01	114.9	58.58
			34.88	114.4	50.81			

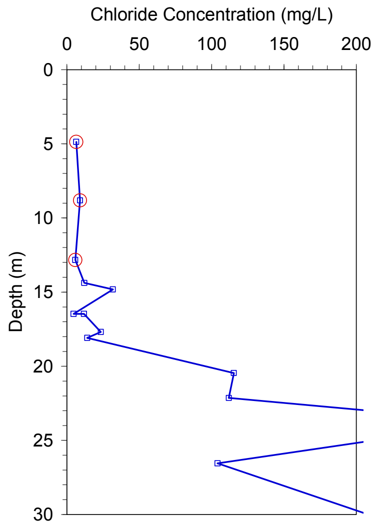

Figure B.17. Pore Water Chloride Concentration Profile for Borehole C3830. Circles identify sample depths used in estimating recharge.

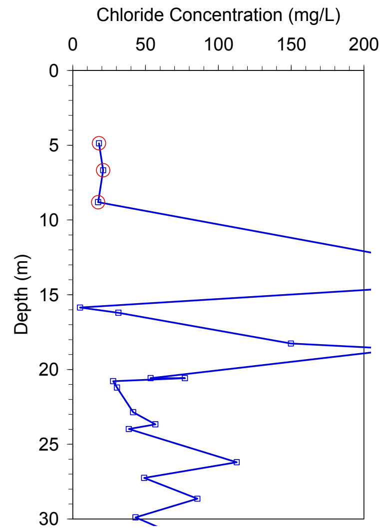

Figure B.18. Pore Water Chloride Concentration Profile for Borehole C3831. Circles identify sample depths used in estimating recharge.

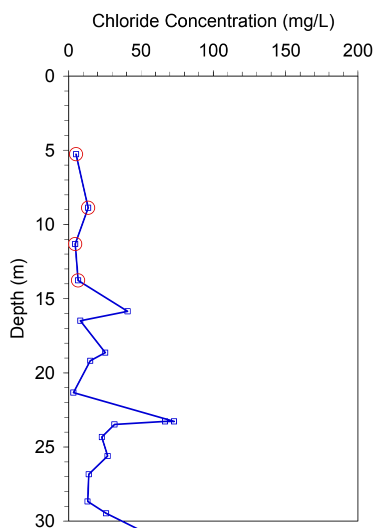

Figure B.19. Pore Water Chloride Concentration Profile for Borehole C3832. Circles identify sample depths used in estimating recharge.

B.3.3.4 PNNL-14849

The borehole data package for WMA T borehole C4104 (Serne et al. 2004b) provides pore water chloride data that can be used to assess recharge. Borehole C4104 is located in the WMA T, between SSTs T-105, -106, -108, and -109. The borehole is closest to SST T-106, being approximately 5.77 m (18.9 ft) from SST T-106. C4104 was drilled in April and May 2003.

Table B.13 gives the pore water chloride concentrations for borehole C4104. The pore water chloride concentration profile for borehole C4104 is presented in Figure B.20. Tank T-106 was confirmed to have leaked in 1973 with documented leaks also from T-103. Pore water chemistry suggested that tank fluid contacted sediment from the depth of 12.2 m (40 ft) to the base of the borehole. In addition, the chloride concentration increases sharply at a depth of 11.40 m (37.4 ft). To avoid possible contamination effects, the recharge analysis was limited to samples collected above 9.37 m (30.7 ft). Using the average chloride concentration of 6.30 mg/L in this depth interval, Equation (B.1) yielded an estimated recharge rate of 6.10 mm/yr.

Table B.13. Pore Water Chloride for Borehole C4104

Borehole C4104					
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
4.93	16.2	5.83	23.17	76.0	53.56
6.92	22.7	4.04	24.70	81.0	40.39
9.37	30.7	9.02	26.62	87.4	20.91
11.25	36.9	557.09	28.23	92.6	19.40
11.40	37.4	29.61	28.52	93.6	25.30
12.20	40.0	28.51	28.86	94.7	36.94
14.12	46.3	110.17	30.39	99.7	32.69
14.32	47.0	21.96	30.59	100.4	41.65
17.80	58.4	87.09	30.89	101.3	22.28
18.01	59.1	27.61	31.08	102.0	14.92
19.32	63.4	119.88	32.12	105.4	34.60
19.51	64.0	34.7			

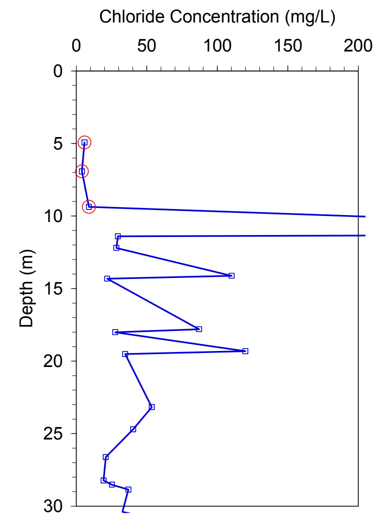

Figure B.20. Pore Water Chloride Concentration Profile for Borehole C4104. Circles identify sample depths used in estimating recharge.

The matric potential data from borehole C4104 were high only in the upper 12 m. In that zone, matric potential values ranged from −0.03 to −0.66 MPa, with an overall average of −0.3. Above 12 m, the high values are indicative of draining conditions relative to the water table location at 69 m. Below 12 m, the values are mixed. Matric potential values ranged from −0.06 to −11 MPa, with an overall average of −2.7. It is unclear how to interpret these data given the extremes in potential values.

B.3.3.5 PNNL-15503

Brown et al. (2006) reported pore water chemistry data for samples from borehole C4297 in WMA C. Borehole C4297 was drilled in February and March 2004 between SSTs C-104 and C-105. While no dates are stated in the report, an overfill of SST C-105 and leakage from subsurface transfer lines occurred, likely contributing waste liquid to the soil at the location of C4297. Introduction of the waste liquid to the soil would have occurred prior to 1974 when contaminated soil near the tanks was first identified. In addition, a liquid level drop in SST C-105 between 1963 and 1967 indicate a leak from SST C-105 likely occurred during that time. The report finds that tank waste–related contamination exists between 0.76 and 48.77 m (2.5 and 160 ft), with mobile contaminants ^{99}Tc and nitrate from the base of tank C-105 to 48.77 m (160 ft).

Table B.14 lists the pore water chloride concentrations as a function of depth for borehole C4297. Figure B.21 shows that the chloride profile is relatively uniform down to a depth of 11.69 m (38.4 ft), with concentrations ranging from 8.23 to 5.08 mg/L. A chloride concentration peak occurs at 11.74 m (38.5 ft), which corresponds to the approximate depth of the base of SST C-105. Therefore, only the eight chloride data values above 11.7 m were used to estimate recharge. Three of those values were below the quantification limit; for this report, the concentration was set at the quantification limit. Using the average chloride concentration of 6.31 mg/L in this depth interval between 2.13 and 11.7 m (7.0 and 38.4 ft), Equation (B.1) yielded an estimated recharge rate of 6.09 mm/yr.

The matric potential data from borehole C4287 were high. In the top 30 m of the boreholes, matric potential values ranged from −0.003 to −0.35 MPa, with an overall average of −0.024. Such high values are indicative of draining conditions relative to the water table location at 76 m.

Table B.14. Pore Water Chloride for Borehole C4297

Borehole C4297					
Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)	Depth (m)	Depth (ft)	Pore Water Chloride (mg/L)
0.76	2.5	7.26	15.62	51.3	12.80
2.13	7.0	8.23	17.43	57.2	8.13
3.66	12.0	7.89	18.75	61.5	9.63
5.18	17.0	6.77	19.05	62.5	15.30
6.71	22.0	5.66	19.17	62.9	12.10
7.86	25.8	6.86[a]	21.02	69.0	8.87
9.45	31.0	4.29[a]	21.72	71.3	16.40
11.34	37.2	5.67[a]	23.59	77.4	10.90
11.69	38.4	5.08	24.25	79.6	12.15
11.74	38.5	36.30	26.97	88.5	25.20
12.44	40.8	8.28[a]	29.44	96.6	12.80
12.56	41.2	6.42	31.70	104.0	13.30
13.34	43.8	15.40	32.32	106.1	25.30
13.87	45.5	8.6[a]	34.81	114.2	20.60
13.98	45.9	9.52	37.98	124.6	25.70

(a) Sample below detection limit; used detection limit of instrument.

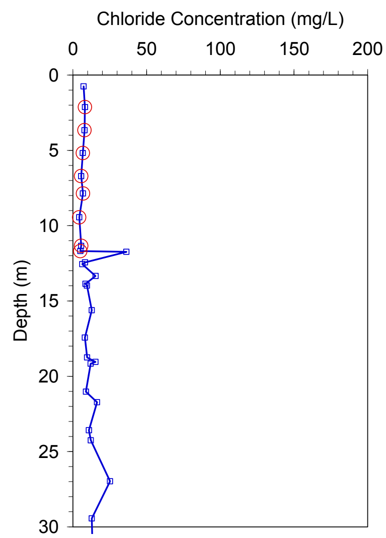

Figure B.21. Pore Water Chloride Concentration Profile for Borehole C4297

B.3.4 Borehole Data Evaluation Summary

Tables B.15 and B.16 summarize the CMB-based estimates of recharge using data from the borehole data packages. Estimated recharge rates under surface-disturbed conditions ranged from 1.6 to 4.0 mm/yr and averaged 2.9 mm/yr. Estimated recharge rates under deeply disturbed conditions ranged from 2.0 to 6.1 mm/yr and averaged 4.6 mm/yr. Overall, the estimated rates under deeply disturbed conditions were about 60% higher than when only the surface was disturbed.

If the estimated recharge rates in Tables B.15 and B.16 are accurate, the assumptions that meteoric water had flushed the backfill material since construction and established an equilibrium condition would be invalidated. For example, assume the following:

- Fifty-five years elapsed between when the backfill material was emplaced and when sediment samples were collected.
- The backfill material has a volumetric water content of 0.10 cm^3/cm^3.
- Drainage through the backfill material is 6 mm/yr.

Given those assumptions, the water percolating downward since construction would have reached a depth of only 3.3 m (10.8 ft). Under this scenario, nearly all of the chloride data used to estimate recharge would be below the meteoric water infiltration front, thus violating the assumption of equilibrium. Reducing the assumed volumetric water content to 0.05 cm^3/cm^3 extends the infiltration front depth to only 6.6 m (21.7 ft), which would still be shallower than most of the pore water chloride samples. As discussed next, we suspect that the estimated rates in the tables are too low.

The estimated recharge rates in Tables B.15 and B.16 are unexpectedly low compared to measurements in similar situations (i.e., coarse sediment and no vegetation) that showed much higher rates. For example, Gee et al. (2005) described a 7.6-m-deep sand-filled lysimeter midway between the Fast Flux Test Facility and the 300 Area. During the 26-year monitoring period, the surface of the lysimeter was kept bare of vegetation to mimic unvegetated conditions such as found around the 200 Areas. Gee et al. reported the average drainage was 62 mm/yr. Fayer and Szecsody (2004) reported an average drainage rate of 113 mm/yr over a 9-year period for an unvegetated 1.5-m-deep lysimeter containing sandy gravel at the Field Lysimeter Test Facility. As an additional comparison, drainage was estimated using the soil texture drainage model of Gee et al. (2005) and grain-size data from Figure 3 of Smoot et al. (1989). The grain-size data were determined for two samples collected from tank farm surfaces in S and U farms in 1989. Based on those data, the estimated drainage rates are 44 and 35 mm/yr, respectively.

Table B.15. Chloride Mass Balance-Based Estimates of Recharge for Surface-Disturbed Sediment (estimates limited to two significant digits)

Borehole	Report Number	Vegetation Condition	Average Pore Water Chloride (mg/L)	Recharge Estimate (mm/yr)
299-W22-48	PNNL-13757-1	No Plants	9.89	3.9
299-W22-50	PNNL-13757-1	No Plants	12.96	3.0
299-E33-338	PNNL-14121	No Plants	21.27	1.8
299-W10-27	PNNL-14594	No Plants	9.70	4.0
299-E27-22	PNNL-15503	No Plants	23.56	1.6
				Average =2.9

Table B.16. Chloride Mass Balance-Based Estimates of Recharge for Deeply Disturbed Sediment

Borehole	Report Number	Vegetation Condition	Average Pore Water Chloride (mg/L)	Recharge Estimate (mm/yr)
299-W23-19	PNNL-13757-2 PNNL-13757-3	No Plants	9.40	4.1
299-W23-234	PNNL-13757-2 PNNL-13757-3	No Plants	7.20	5.3[(a)]
299-E33-45	PNNL-14083	No Plants	17.17	2.2
C3830	PNNL-14594	No Plants	7.09	5.4
C3831	PNNL-14594	No Plants	18.83	2.0
C3832	PNNL-14594	No Plants	7.40	5.2
C4104	PNNL-14849	No Plants	6.30	6.1
C4297	PNNL-15503	No Plants	6.31	6.1[(b)]
				Average =4.6

(a) One of two samples was below quantification limit; reset to limit.
(b) Three of eight samples were below quantification limit; reset to limit.

Three of the tracer studies discussed in Section B.2 relied on samples from one of the boreholes examined in this section, i.e., 299-W22-48. Maher et al. (2003) used the strontium isotope ratio technique to estimate a recharge rate of 7 ± 3 mm/yr. Maher et al. (2006) used the strontium technique along with a uranium isotope technique to refine the recharge estimate to 5 ± 2 mm/yr. In both cases, the estimated rates are similar to the chloride-derived rates of 3.9 mm/yr. As discussed above, these rate estimates are much lower than expected based on lysimeter monitoring data.

The third relevant tracer study was by DePaolo et al. (2004), who used the ^{18}O estimation technique to estimate a recharge rate of 35 to 60 mm/yr recharge. This estimate is much higher than the other tracer-based estimates and is much closer to expectations based on monitoring data. This technique bears further consideration as a tool for estimating recharge rates in tank-farm–like settings.

The matric potential data are indicative of draining profiles relative to the local water table depths. Hydraulic property data could be used to predict recharge rates directly, but the variability in properties would likely result in a very large range of recharge predictions that would not allow us to make sense of the chloride-based estimates.

The CMB method is well-documented and used around the world to estimate recharge rates, but as this study shows, the method may not always be applicable. Potential reasons why recharge rates estimated using the observed chloride pore water profiles do not agree with anticipated recharge rates include

- measurement limitations – Under the high recharge conditions anticipated for disturbed and unvegetated sites, the pore water chloride concentrations will be low, which means the extract concentrations will be even lower, quite possibly at or near the quantification limit of the measurement technique (Gee et al. 2005). Under this situation, analytical limitations or contamination of the sample (e.g., mineral dissolution, sweat, sampling containers) could significantly alter the results. Figure B.22 shows how sensitive recharge rate estimates are to extract concentration and water content.

- backfill disequilibrium – For the deeply disturbed sites, the chemical conditions in the backfill may not have equilibrated completely since construction, and what is being observed is the chloride signature of the equilibrating backfill material.

- incorrect chloride deposition rate – Chloride deposition rates used to estimate recharge for this report are based on deep chloride profiles representing time spans of a thousand or more years. Modern chloride deposition at Hanford has not been measured. Uncertainty in the chloride deposition rate at Hanford potentially introduces large error in recharge estimates derived using CMB. Long-term efforts are needed to quantify chloride deposition rates and their variability throughout Hanford. Fayer and Szecsody (2004) suggested that chloride could have emanated from coal-fired power plants and other facilities at various locations throughout the Hanford Site during the period of active Hanford operations. Measurements made today would not be able to reveal what was being deposited while facilities like the coal plants operated. Additional analysis is needed at sites throughout the 200 Area plateau that may have received elevated chloride deposition due to coal combustion emissions or chemical separation operations.

- Mineral dissolution – The strontium and uranium isotope methods both depend on knowledge of dissolution rates. The chloride method may be affected by mineral release of chloride during oven drying of the sample prior to analyzing for chloride (Gee et al. 2005).

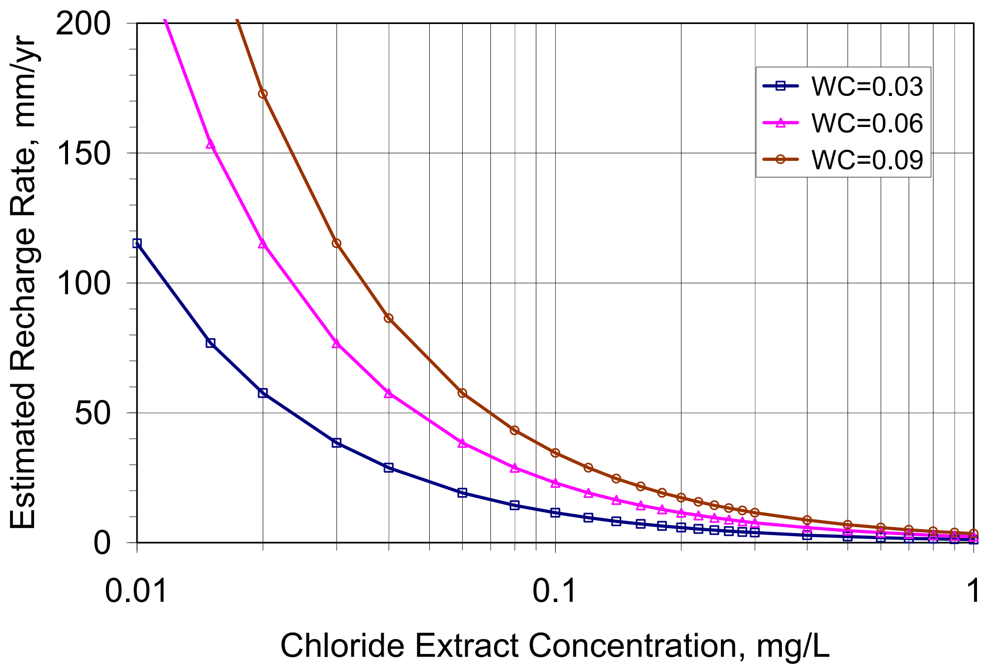

Figure B.22. Sensitivity of Recharge Rate Estimate to Extract Concentration and Water Content (WC = water content, g/g). For reference, the average annual precipitation is 173 mm/yr. Dilution is 1:1.

- unknown chloride sources – Chloride may have been added to the sediment via at least three mechanisms. First, chloride may have been present in the crushed gravel and rock that was brought onsite and applied to tank farm surfaces as part of the controlled, clean, and stable program. Second, chloride may be a degradation product of commercial weed killer that is applied to the surface of tank farms to restrict the growth of vegetation. Among the herbicides applied is diuron,[a] the chemical structure of which includes chloride. Third, undocumented waste leaks could have contained chloride. While possible, undocumented waste leaks would be expected to leave a signature of other contaminants, a signature that was not mentioned in the borehole data packages.

For situations in and around tank farms, where recharge estimates are desperately needed yet sorely lacking, the CMB method offered an opportunity to leverage existing data from the borehole data packages into estimates of recharge rates. Unfortunately, the applicability of the CMB method to tank farms is hampered by multiple factors that include the lack of documentation of the surface and vegetation conditions prior to sampling, lack of continuous sampling in the upper 15 m, uncertainty regarding measurement resolution, and questions about the chloride deposition rate. Because the recharge estimates in Tables B.15 and B.16 are inconsistent with measured rates in well-defined lysimeter experiments, we conclude that the CMB method does not provide defensible estimates of recharge rates using the data available in the existing borehole data packages. Therefore, we recommend against using the rates in Tables B.15 and B.16 to represent recharge in tank farms.

B.4 Recommended Recharge Rates

The suite of available tracer estimates of recharge was reviewed to provide the recommendations in Table B.17. When there was uncertainty about soil type, both types are indicated. For all of the measurements in natural settings involving sandy soil, the estimates are consistently in the range of 0 to 6 mm/yr. Even the relatively new strontium-based method yielded a rate of 5 mm/yr. Although the strontium method is listed in Table B.17 as being applied to a graveled surface, the analysis is of the deeper sediment and thus reflective of conditions 100s to 1000s of years ago rather than current surface conditions. Overall, what these results suggest is that undisturbed soil and plant communities at the Hanford Site are capable of minimizing recharge. From an ecological standpoint, that is consistent with maximizing the use of a scarce resource, i.e., water.

In contrast to the low rates in natural settings, the single tracer measurement in a disturbed area with no plants yielded a recharge estimate of 35 to 60 mm/yr. That rate is consistent with lysimeter measurements of similar surfaces. Application of this method is fairly new at Hanford, but its ability to estimate recharge in disturbed areas is promising and may offer a valuable companion to other tracer techniques such as chloride mass balance.

One of the significant challenges in assembling Table B.17 was the inability to assign specific soil types to specific tracer tests. As the experience at the IDF Site shows, each mapped soil unit can have significant variability within itself and the "boundaries" between soil types may be artificial. The soil profile at each tracer test ought to be characterized by a soil scientist and photo-documented so that the tracer results can be grouped according to actual soil conditions rather than the soil map.

(a) Personal communication from David Myers, CH2M HILL Hanford Group, Inc., April 10, 2007.

A related challenge in assembling Table B.17 was that characterization of the plant community was not very quantitative. One researcher's mature shrub community may be another's sparse grasses and shrubs. The community at each tracer test ought to be characterized by a plant ecologist and photo-documented. This characterization should be done prior to any site preparation work to install drill pads.

Each tracer method is based on a suite of assumptions that adds uncertainty to the estimate of recharge. Some of the assumptions are tracer-specific; some are common to all tracers. By far, the CMB method is the one most frequently used for estimating recharge at Hanford. Therefore, uncertainties in the CMB method can have significant implications for Hanford assessments. The review of the published recharge analyses and borehole data packages highlighted several uncertainties that exist with respect to application of the CMB method at Hanford, including deposition rate, location history, sampling method and frequency, and chloride extraction and analysis procedure (including gravel content, water content, dilution ratio, and quantification limit). A comprehensive study is needed to better identify the limitations of the CMB method in estimating recharge in high recharge conditions and confirming the uncertainties in low recharge conditions.

The lessons learned during preparation of this appendix show that more effort in planning and characterization would significantly enhance the value of the recharge estimate, both in reducing the uncertainties and in providing corroborative data. These lessons should be considered in any future endeavors to estimate recharge.

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Table B.17. Tracer-Based Estimates of Recharge Rates in and Around the 200 Areas. The precision of the recharge estimates was limited to 2 significant digits. Values in parentheses are from the original sources.

Borehole	Reference	Soil Type	Vegetation Condition	Tracer Method	Recharge Estimate (mm/yr)
Yakima Barricade (3 boreholes)	Murphy et al. (1996)	Ba or Wa (similar to Qu)	Mature shrub-steppe	Cl	0.011, 0.012, 0.013
Wye Barricade	Murphy et al. (1996)	Rp	Grass with scattered shrub	Cl	4.0 (4.01)
B10	Prych (1998)	Ba or E1	"Sagebrush with sparse grass"	Cl	2.8
B12	Prych (1998)	Ba or E1	"Sagebrush with sparse grass"	Cl	5.5
B17	Prych (1998)	Ba or E1	"Sagebrush with sparse grass"	Cl	1.8
B18	Prych (1998)	Ba or E1	"Sagebrush with sparse grass"	Cl	1.2
B20	Prych (1998)	Ba	"Sagebrush with sparse grass"	Cl	0.66
B20	Prych (1998)	Ba or E1	"Sagebrush with sparse grass"	^{36}Cl	<2.6
E24-161	Fayer and Szecsody (2004)	Ba[(a)]	Mature shrub-steppe	Cl	0.16
E24-162	Fayer and Szecsody (2004)	Rp[(a)]	Mature shrub-steppe	Cl	0.71
E17-21	Fayer and Szecsody (2004)	Rp[(a)]	Mature shrub-steppe	Cl	1.0 (1.01)
8501	Fayer and Szecsody (2004)	Rp[(a)]	Mature shrub-steppe	Cl	1.1 (1.11)
B8502	Fayer and Szecsody (2004)	RP[(a)]	Mature shrub-steppe	Cl	2.1 (2.11
E24-21	Fayer and Szecsody (2004)	Ba[(a)]	Mature shrub-steppe	Cl	0.24
E17-22	Fayer and Szecsody (2004)	RP[(a)]	Mature shrub-steppe	Cl	0.62
299-W2-48	DePaolo et al. (2004)	Gravel cover	NV	^{18}O	35 to 60
299-W2-48	Maher et al. (2006)	Gravel cover	NV	Sr and U isotopic ratios	5 ± 2
699-50-59	Keller (2007)	Ba or E1	Mature shrub-steppe	Cl	2.5

(a) Fayer and Szecsody (2004) recommended treating the soil at the IDF as a single soil type.
Soil abbreviations: Ba = Burbank loamy sand, E1 = Ephrata sandy loam, Rp = Rupert sand, He = Hezel sand, Qu = Esquatzel silt loam, GC = gravel cover
Vegetation conditions: ss = shrub-steppe; b = blended; sr = shallowrooted; nv = no vegetation.

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Appendix C

Simulation Estimates of Recharge Rates

Appendix C

Simulation Estimates of Recharge Rates

This appendix provides a set of simulation-based estimates of recharge rates to support the forthcoming RCRA Facility Investigation (RFI) report for Hanford SST WMAs. Direct measurements of recharge are not available for past and future conditions in the WMAs, nor are they available for many current conditions. Therefore, computer simulations are used to estimate rates for those periods and conditions for which data do not exist or are incomplete.

The scope of this appendix includes a review of previous simulation work and an update to some previous simulation results using the entire record (1957 through 2006) of hourly weather data from the Hanford Meteorological Station. The specific simulations include the base case from Smoot et al. (1989), the five 200 Area soil types analyzed by Fayer and Walters (1995), and a new simulation that uses the backfill properties reported in the WMA S-SX FIR (CH2M HILL 2002). These simulation examples are illustrative of recharge conditions in and around tank farms but should not be considered definitive for any specific farm, given the lack of site-specific soil property data.

Some of the material in this appendix was taken from the various predecessor simulation reports, including Smoot et al. (1989), Fayer and Walters (1995), and Fayer and Szecsody (2004).

C.1 Previous Hanford Recharge Simulation Studies

Smoot et al. (1989) used numerical simulations to evaluate the movement of a contaminant plume that resulted from a 1973 leak from tank T-106. To support that evaluation, Smoot et al. used an early version of the UNSAT-H computer code to simulate infiltration rates. Their base case was a simulation of recharge for a non-vegetated backfill material for the period 1947 to 2010. They used weather data for 1947 and the period 1957 to 1988. For the remainder of the time, they used a weather generator computer code to estimate precipitation, maximum and minimum air temperature, and solar radiation. For dew point temperature, they used weekly averages. For wind speed and cloud cover, they used monthly averages. During the simulation, the suction head of the surface node was not allowed to exceed 500 cm. This restriction was necessary because the version of UNSAT-H used for the study was single precision, which made it difficult to solve for the highly nonlinear conditions that occur as the soil surface dries out.

The average drainage for the base case (called Case 1 by Smoot et al. 1989) was 130.8 mm/yr, which represented 77% of the average 168.8 mm/yr precipitation received. Annual drainage varied between 70.6 and 225.2 mm (standard deviation = 30.3 mm), reflecting the dependence on weather conditions.

Smoot et al. (1989) also examined some parameter sensitivities. For a 10-year period, they showed that lowering the saturated conductivity by a factor of 10 reduced drainage by about 8%; a value 10 times higher increased drainage by about 13%. Another variable examined was the addition of either silt loam or clean gravel to the backfill surface. For the 10-year period examined, the results showed that a 15-cm-thick silt loam layer reduced drainage to less than 1 mm within 5 years. In contrast, the 15-cm-thick clean gravel increased drainage to nearly 96% of the precipitation. In the concluding section, Smoot et al. noted that no direct measurements of hydraulic properties had yet been made of tank farm sediments and that such measurements were necessary to substantiate the simulation results.

Gee et al. (1992) used the UNSAT-H code to examine water storage (a precursor to downward water movement) in the 200 East Lysimeter from 1972 to 1985. A simulation without plants showed that water storage should have increased significantly during that period. There were no records to document plant cover on the lysimeter, but some pictures taken in 1973 suggested that plants were present on the surface. By including plants in the simulation, the predicted water values more closely matched the water storage measured in 1985, thus suggesting the importance of plants in estimating recharge.

Fayer and Walters (1995) created a map of recharge rates for the Hanford Site. They used a combination of chloride tracer data, lysimeter and water content measurements, and simulations (using UNSAT-H) to estimate recharge rates. They focused their efforts on natural diffuse recharge and did not address leaking tanks, roads, or other structures. They used the Hajek (1966) soil map and a recent vegetation map to allocate the recharge estimates to specific soil-vegetation combinations. Fayer and Walters treated all disturbed areas as having no plants and having the soil type originally mapped by Hajek (1966). In reality, the soil surfaces of many of those areas were disturbed and graveled, and the impacts of such changes were not considered in the report.

Rockhold et al. (1995) conducted simulations of recharge to highlight effects of soil type, layering, and vegetation type to support the performance assessment for the low-level waste facility (now known as the Integrated Disposal Facility, or IDF) proposed for the 200 East Area. Using the 30-year weather record from 1963 to 1992, they simulated a sand, a silt loam, and a 30-cm-thick silt loam layer above sand (i.e., a capillary break design). For each soil, they examined the case without plants as well as cases with cheatgrass, bunchgrass, and shrubs. Their simulation results showed that the silt loam reduced recharge by 50 to 65% (depending on vegetation) relative to the sand. When the silt loam was layered above sand, the recharge was reduced by 80% relative to the silt loam alone. With respect to vegetation, the results showed a significant reduction in recharge as plants with deeper root systems were simulated. Relative to the recharge predicted in the absence of plants, adding cheatgrass reduced recharge 20 to 30%, bunchgrass reduced it 60 to 75%, and shrubs reduced it 87 to 95%. The authors extended those results to the potential effects of fire, which would remove the deep-rooted shrubs in favor of bunchgrass and cheatgrass, with the likely outcome that recharge rates would increase (relative to rates under shrubs).

In their simulation results, Rockhold et al. (1995) observed a distinct time lag between significant recharge periods and cautioned against using short monitoring periods. They stated that "…referring to the mean annual recharge rate is really only appropriate if the mean is taken over a sufficiently long period containing a statistically significant number of extreme events." The duration of time that would be sufficient is unclear, but their results showed that duration must be longer when the recharge rate is lower. In the context of soil and vegetation, a silt loam would require more time than a sand, and a shrub would require more time than cheatgrass.

Fayer et al. (1999) simulated recharge rates to support the 2001 Immobilized Low-Activity Waste (ILAW) Performance Assessment (Mann et al. 2001). The results in Table C.1 show that the surface barrier design (1.0-m-thick silt loam layer above gravel) limited recharge to less than 0.1 mm/yr, which is much better than the design goal of 0.5 mm/yr. The cover maintained this performance level when plants were removed, when the climate became wetter and cooler, and when 20 cm of the silt loam layer were eroded. The cover also maintained this performance when windblown sand was deposited, but significant drainage (32.7 mm/yr) occurred when plants were removed. Under a wetter, cooler climate, the presence of dune sand on the cover resulted in 16.9 mm/yr of annual drainage, even though shrub-steppe vegetation was present.

Table C.1. Simulated Long-Term Drainage Rates Using the Isothermal, Non-Hysteretic Mode of UNSAT-H, a 41-Year record of Meteorological Data, and a Shrub-Steppe Plant Community (unless noted otherwise) (Source: Fayer et al. 1999)

		Simulated Long-Term Drainage Rates (mm/yr)				
Variable	Condition	Modified RCRA Subtitle C Barrier	Rupert Sand	Burbank Loamy Sand	Dune Sand on Barrier	Eroded Surface Barrier
Climate[a]	Current (1957 to 1997)	<0.1	2.2	5.2	<0.1	<0.1
	P↓	NA[b]	<0.1	NA	NA	NA
	P↑	NA	13.2	NA	NA	NA
	P↓T↑	NA	<0.1	NA	NA	NA
	P↓T↓	NA	<0.1	NA	NA	NA
	T↓	NA	7.5	NA	NA	NA
	T↑	NA	0.6	NA	NA	NA
	P↑T↑	NA	5.2	NA	NA	NA
	P↑T↓	<0.1	27.0	36.8	16.9	<0.1
Vegetation	Cheatgrass	NA	33.2	NA	18.4	NA
	No plants	<0.1	44.3	52.5	32.7	<0.1
	No plants, future climate (P↑T↓)	NA	88.6	98.0	NA	NA
Shrub leaf area index	High (0.4 vs. 0.25)	NA	1.6	NA	NA	NA
	Low (0.1 vs. 0.25)	NA	5.6	15.2	4.1	NA
Rupert sand properties	Higher $K(h)$ vs. Rupert sand	NA	2.7	NA	NA	NA
	Lower $K(h)$ vs. Rupert sand	NA	3.3	NA	NA	NA
Complete areal plant coverage	Cheatgrass	NA	26.6	NA	NA	NA
	Shrub	NA	<0.1	NA	NA	NA
Irrigation efficiency	75%	26.4	58	NA	NA	NA
	100%	<0.1	30	NA	NA	NA

(a) Climate change was represented using changes in precipitation (P) and temperature (T). An increase is represented by ↑ and a decrease by ↓. The ranges were 50 to 128% of modern P and -2.5 to 2.8°C of modern T.

(b) NA = Not analyzed.

Drainage rates for the Rupert sand and Burbank loam sand soils were 2.2 to 5.2 mm/yr under shrub-steppe vegetation. Additional simulations highlighted sensitivities to variations in climate, soil hydraulic properties, plant parameters, and irrigation.

The authors of the S/SX FIR (CH2M HILL 2002) examined the influence of high tank temperatures (often in excess of 100°C) on recharge. The expectation was that the high temperatures would increase evaporation rates and thus reduce recharge. For the analysis, the authors fixed the recharge rate at 100 mm/yr, the surface temperature at 12.8°C, and the surface relative humidity at 30%. The quantity of water vapor that moved to the surface was attributed to the increased temperature. When expressed as a fraction of the imposed recharge rate of 100 mm/yr, the results suggested that temperature increases caused by the tanks increased evaporation by no more than 2%. This result is surprising, given that other sections of the FIR discuss significant thermal effects up to 23 m below the tanks. Most likely, the imposition of fixed conditions at the surface affected the analysis. The authors concluded that, had the ground-surface temperature been computed using a radiative-convective heat transfer approach, the percentage of meteoric recharge lost through the ground surface would have been higher.

Fayer and Szecsody (2004) conducted simulations to augment the work of Fayer et al. (1999) for the IDF. They updated the simulation results with six additional years of meteorological data (1998 through 2003). They evaluated the impact of a deeper simulation domain, hysteresis, and natural heat flow. They also examined the impact of variability on the silt loam hydraulic properties.

The results confirmed that the three surface barrier scenarios showed less than 0.1 mm/yr drainage, with or without vegetation, thus demonstrating the robustness of the surface barrier. Noteworthy is that the simulation period included some extreme events, including the 24-hr record precipitation of 48.5 mm. This amount is nearly equivalent to the predicted 1,000-yr 24-hr amount of 51.6 mm (Hoitink et al. 2005). The two soil scenarios showed some sensitivity to the additional six years of weather data. Under shrub-steppe covers, the predicted recharge rates were 7.7 to 18% less. Without plants, the rates were only 1% less. In both cases, the rates dropped about 0.4 mm/yr. Because the rates under shrub-steppe conditions were low already, the percentage was higher.

The set of simulations without plants was repeated with a 6–m-deep profile (2 m deeper than Fayer et al. 1999) to demonstrate sensitivity to domain size. An additional simulation was conducted with a 10-m-deep profile to examine sensitivity to the bottom boundary temperature condition. The results of all the deeper profiles were essentially identical to the results for the shallower profiles, indicating that a 4-m-deep domain is adequate for simulations such as these.

The simulations with hysteresis indicated that the effect on recharge is sensitive to soil type and the presence of plants. In the presence of shrubs, recharge remained less than 0.1 mm/yr beneath the surface barrier (1.0 m silt loam) and the eroded barrier (0.8 m silt loam) but increased beneath the surface barrier covered by dune sand (from <0.1 to 0.7 mm/yr). For the soils, the results were mixed. Recharge for Rupert sand increased 1.8 to 5.7 mm/yr, whereas the rate for Burbank loamy sand decreased from 4.8 to 3.9 mm/yr. In the absence of shrubs, the surface barrier and eroded barrier still limited recharge to less than 0.1 mm/yr, but the recharge rate beneath the surface barrier with dune sand (and no vegetation) nearly doubled from 32.3 to 62.6 mm/yr. Hysteresis increased recharge in the two soil scenarios as well, but not by as much.

The simulation with heat flow resulted in 16% less recharge. The reduction was due to the increased evaporation rate. The representations of evaporation in the heat flow and isothermal versions of the code are different and could account, in part, for the differences in evaporation and recharge.

A set of 16 simulations was conducted to demonstrate the sensitivity of the surface barrier to variations in the hydraulic properties of the upper 0.5-m silt loam layer. The results showed that all 16 manifestations of hydraulic properties yielded the same zero-drainage outcome, once again highlighting the robustness of the surface barrier design. Six of the simulations generated runoff, but the amounts were small. Five averaged less than 0.1 mm/yr of runoff; the sixth averaged 0.3 mm/yr.

C.2 Methods

PNNL used the one-dimensional numerical code UNSAT-H Version 3.01 to estimate recharge rates for conditions on the Central Plateau. Five natural soil types (Hajek 1966) and one unique Hanford soil surface condition (i.e., a graveled surface) cover the 200 Areas. Plant communities are not allowed within the SST WMAs. In the area surrounding the SST farms, the plant communities range from nonexistent to relatively undisturbed shrub-steppe. SST farms are relatively flat, but some have edges that are protected with steep unvegetated side slopes composed of sandy gravel.

Based on this information, seven scenarios were identified for simulation—five soil types and two SST backfill materials. Given that the dominant natural processes are vertical and that areally averaged recharge is being estimated, the one-dimensional UNSAT-H code is appropriate. For situations in which site-specific estimates are required and in which multidimensional features, events, or processes may significantly affect recharge, a multidimensional simulation may be needed to estimate recharge rates. The simulation cases and associated parameters are described in the following sections.

C.2.1 Simulation Cases

Nine sets of simulations were conducted to evaluate recharge. For each set, three vegetation conditions were evaluated and yielded a total of 24 unique simulations. All 24 simulations were conducted using the 50-year weather record (1957 through 2006) from the Hanford Meteorological Station. The first five sets of simulations involved extending the Fayer and Walters (1995) simulations of the five soil types in the 200 Areas to include the years 1993 through 2006. These simulations were conducted because they are appropriate for representing much of the area surrounding the 200 West and East Areas, and some areas inside as well, that still remain relatively undisturbed. They also serve as estimates of what currently disturbed areas might one day become.

The sixth set of simulations was a repeat of Case 1 of Smoot et al. (1989). This case was conducted with an older version of UNSAT-H and needed to be redone. This set provided an opportunity to replace the estimated meteorological parameters with 50 years' worth of measured data, and it allowed us to analyze how recharge for this specific soil would change as plants were introduced.

The seventh set of simulations was a modified version of the Smoot et al. (1989) case. As explained in more detail in Section C.2.4, this set of simulations was conducted using a modified form of the hydraulic functions to overcome a limitation inherent in the Smoot et al. analysis.

The eighth set of simulations used the backfill properties used in the FIR for the WMA S-SX (CH2M HILL 2002) and all subsequent FIR documents to date.

For each of the first eight sets of simulations, three vegetation conditions were evaluated—shrub cover, cheatgrass cover, and no plants.

The ninth and final set of simulations was conducted with the bottom boundary at different depths to determine how shallow the boundary could be before it started to impact the recharge estimates.

C.2.2 Code Description

Simulations were conducted using the UNSAT-H computer code (Fayer 2000). UNSAT-H was accepted for use at Hanford via the Tri-Party Agreement process (DOE 1991). The IDF (formerly ILAW) project has used this code since 1995 specifically to calculate recharge rates.

The UNSAT-H code has been tested with lysimeter data. Fayer et al. (1992) and Martian (1994) compared predicted and measured water storage values for lysimeters at Hanford. Both found that calibration of several parameters improved the match of predicted to measured values as determined by the root-mean-square (RMS) error. For a 1.5-year test of a lysimeter receiving an enhanced precipitation treatment, Fayer et al. (1992) calculated a RMS error of 0.8 cm after calibration (versus 2.2 cm without calibration). Martian (1994) looked at a much longer time period (5.5 versus 1.5 years) and found the RMS error was higher—about 1.8 cm for the calibrated model. The analysis was not done for the uncalibrated model. Martian determined the correlation coefficient for the comparison of measured and simulated soil water storage was 0.94, which is quite good.

Fayer and Gee (1997) extended the comparison of Fayer et al. (1992) to 6 years. The data were collected from a nonvegetated weighing lysimeter containing 150 cm of silt loam over sand and gravel. They found that the RMS error for water storage predictions was about 2.3 cm, regardless of whether the model was calibrated or whether it included heat flow or hysteresis. Fayer and Gee (1997) extended the comparison to matric potential and drainage. They found that the simulation with hysteresis was far better at predicting matric potentials throughout the 6-year period, and it was the only simulation to predict drainage (52% of the measured amount, with timing that matched the observations).

As discussed in Section C.1, Gee et al. (1992) used UNSAT-H to demonstrate how plants could explain the measurements made in the 200 East Lysimeter. Rockhold et al. (1995) followed that with a simulation exercise in which the UNSAT-H prediction of lysimeter water storage after 14 years agreed with measured water storage in 1985.

Khire et al. (1997) tested UNSAT-H for simulating water movement in surface barrier test plots in a semiarid setting in Washington and a humid setting in Georgia. They tested the model using a 3-year record of data that included overland flow, soil water storage, evapotranspiration, and percolation. Time-series plots of the data and predictions showed that UNSAT-H generally mimicked the seasonal trends.

Khire et al. (2000) used UNSAT-H to assess the performance of capillary barriers relative to layer thickness, unsaturated hydraulic properties, and climate at four sites in the United States. They concluded that barrier performance was sensitive to all three variables.

Scanlon et al. (2002) compared the ability of seven codes to simulate the performance of engineered barriers in Texas and Idaho. They reported that most of the codes, including UNSAT-H, reasonably reproduced the measured water balance components. They also reported that the weakest comparison was of runoff. As Section 3 of this report attests, surface runoff is a very minor element of the water balance at Hanford and thus should not preclude the use of UNSAT-H.

The UNSAT-H code has been used at the Hanford Site to estimate the areal distribution of recharge rates (Fayer et al. 1996). The code also has been used elsewhere to evaluate infiltration through surface barriers (Magnuson 1993) and surficial sediments (Martian and Magnuson 1994).

C.2.3 Simulation Domain and Discretization

The simulation domains were 4.0 m for the first eight sets of simulations. Fayer and Szecsody (2004) examined larger domains and determined that 4.0 m was sufficient. For the ninth set of simulations, smaller domains of 1.0, 1.5, and 2.0 m were examined.

The node spacing in all simulations was 0.2 cm at the soil surface and gradually increased with depth to no more than 30 cm. At material interfaces, the node spacing was decreased to 2 cm. Changes in node spacing from node to node were limited to less than 50%.

Time-step sizes were allowed to range from 10^{-10} to 1 hour in accordance with minimizing the mass balance error of any particular time step.

C.2.4 Soil Information

Soil hydraulic properties consist of the soil water retention function and the hydraulic conductivity model. Soil water retention was described with the van Genuchten function, and hydraulic conductivity was described with the Mualem conductivity model. Table C.2 lists the parameters for the materials making up each soil type; these parameters are identical to those used by Smoot et al. (1989), Fayer and Walters (1995), and CH2M HILL (2002). Possible changes in soil hydraulic properties in response to soil development were not addressed.

During the preparation of this report, it became clear that the backfill hydraulic properties reported by Smoot et al. (1989) were extremely nonlinear. When Smoot et al. conducted their simulations, they used an early single-precision version of the UNSAT-H code that was not able to solve the problem when the domain was allowed to dry out to suction heads of 10^6 cm. To remedy that difficulty, they set the upper head limit to 500 cm. With the current version of the code (Fayer 2000), the domain can dry out readily to 10^4 cm, but much more computational effort is required to dry it out further. At a suction head of 10^4 cm, the water content is equivalent to an effective saturation of 0.0002% (compared to 0.053% at a suction head of 500 cm). To provide possibly more realistic representation of hydraulic properties for the Smoot et al. backfill, we used the modified van Genuchten function proposed by Fayer and Simmons (1995). Only three parameter changes were necessary. The suction head h_m represents zero water content; the value was set to 10^7 cm. The value of n was modified using the empirical equation provided by Fayer and Simmons. The value of θ_a was calculated using h_m, the modified n, the remaining parameters in Table C.2, and the suction head-water content pair 15,300 cm and 0.02 (Smoot et al. 1989).

Figure C.1 shows that the water retention properties of the two backfill materials vary substantially. The properties of two soil types also are displayed to show how the backfill properties compare.

Table C.2. Parameters Used to Describe Soil Hydraulic Properties in the Simulations. The van Genuchten parameter *m* was set equal 1-1/*n*. The pore interaction term was specified using the standard value of 0.5.

Soil Type (depth interval, cm)	θ_s cm^3/m^3	θ_r cm^3/m^3	α 1/cm	*n* --	K_s cm/h
Fayer and Walters (1995) Rupert Sand					
Sand (0 to 400)	0.433	0.0381	0.106	1.78	35.3
Fayer and Walters (1995) Burbank Loamy Sand					
Sand (0 to 41)	0.433	0.0381	0.106	1.78	35.3
Loamy sand, 45% gravel (41 to 76)	0.279	0.0160	0.0292	1.35	2.44
Loamy sand, 85% gravel (76 to 89)	0.0760	0.0040	0.0292	1.35	0.519
Sandy gravel (89 to 400)	0.0833	0.0084	0.0061	1.52	0.572
Fayer and Walters (1995) Ephrata Sandy Loam					
Sandy loam (0 to 33)	0.47	0.0426	0.117	1.48	3.2
Sandy loam with 70% gravel (33 to 71)	0.141	0.0128	0.117	1.48	0.592
Sandy gravel (71 to 400)	0.0833	0.0084	0.0061	1.52	0.572
Fayer and Walters (1995) Hezel Sand					
Loamy sand (0 to 41)	0.507	0.0282	0.0292	1.35	6.48
Silt loam (41 to 400)	0.444	0.0	0.011	1.7612	4.32
Fayer and Walters (1995) Esquatzel Silt Loam					
Silt loam (0 to 400)	0.444	0.0	0.011	1.7612	4.32
Smoot et al. (1989) Backfill					
Smoot backfill (0 to 400)	0.2585	0.02	0.1008	2.922	4.46
Smoot et al. (1989) Backfill with Modified van Genuchten					
Smoot backfill (0 to 400)–modified VG[(a)]	0.2585	0.0497	0.1008	3.43	4.46
CH2M HILL (2002) S-SX FIR Backfill					
S-SX FIR backfill (0 to 400)	0.138	0.01	0.021	1.37	2.016
(a) For the modified VG function, the listed θ_r parameter actually represents θ_a and the parameter h_m is 10^7.					

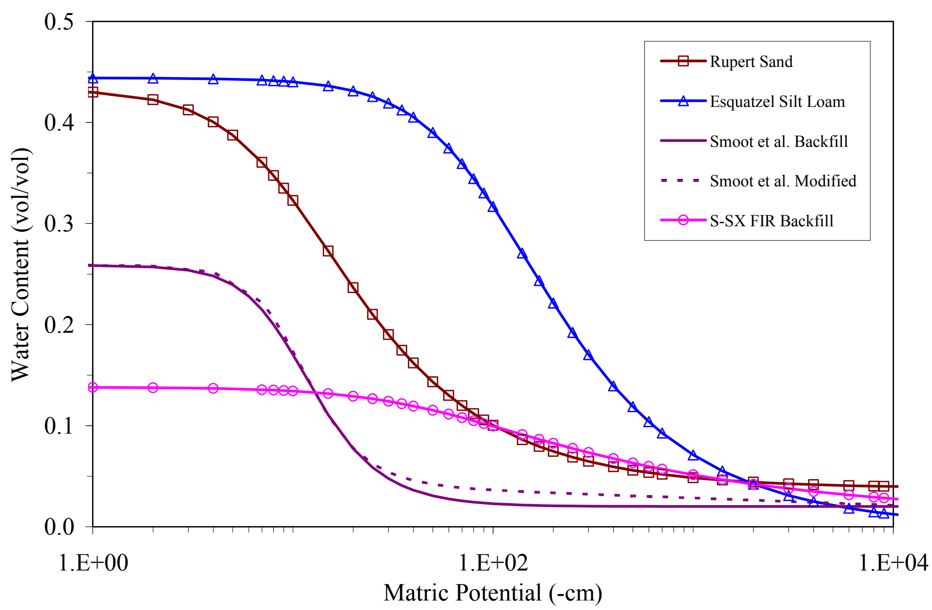

Figure C.1. Water Retention Properties of Backfill Materials and Two Soil Types (Rupert sand and Esquatzel silt loam)

Figure C.1 also shows that the modified properties for the Smoot et al. backfill are very similar to the original, showing only a slight difference in the drier region. As will be shown later, this slight difference has a significant impact on the results.

C.2.5 Initial Conditions

All simulations were started using the weather data for January 1, 1957. Initial matric potential values were not available for any of the scenarios, so the initial values were specified as -10^3 cm for natural soils and −100 cm for soils beneath a gravel surface. These values are wetter than some measured vadose zone potentials (e.g., Prych 1998), so early simulated drainage could reflect this initial water. However, this limitation was overcome by repeating the 50-year sequence until the beginning and ending water storage values were within 0.1 mm of each other. This procedure uncoupled the results from the impact of initial conditions. The implicit assumption is that the 50-year weather record, when repeated, is representative of much longer periods.

C.2.6 Boundary Conditions

Boundary conditions describe the water inputs and outputs at the top and bottom of the simulation domain. For this report, these conditions are the weather data needed to calculate evapotranspiration, the precipitation data, and the drainage rate from the bottom of the profile. The weather and precipitation data were derived from the meteorological data collected at the Hanford Meteorological Station (HMS) for the years 1957 through 2006. The HMS is just outside the northeast corner of the 200 West Area at an elevation of 223 m (Hoitink et al. 2005). SST WMAs span elevations of 200 to 210 m in 200 West and 195 to 215 m in 200 East. More broadly, the elevations in and around the 200 West Area range from 195 to 240 m and in 200 East Area from 175 to 225 m. The HMS elevation differs by less than 30 m from those elevations so that topographic effects on weather were considered to be negligible.

The current climate conditions were represented using the daily weather data. Measured hourly precipitation rates were used to describe the water inputs. Snowfall was treated as an equivalent rainfall at the time it occurred. Weather data such as wind speed, cloud cover, relative humidity, solar radiation, and maximum and minimum air temperature were used to calculate potential evaporation using the Penman Method (Doorenbos and Pruitt 1977).

The bottom boundary was represented with a unit-gradient condition. This condition is generally acceptable when the boundary is well below the deepest plant roots and there are no significant thermal or gas phase gradients. The deepest plant roots are at 2 m, which is well above the bottom boundary at 4 m. The ninth set of simulations examines boundaries shallower than 4 m to determine whether evaporation is affected.

C.2.7 Plant Information

The plant community fills several important ecological roles (e.g., protection from wind and water erosion; nutrient recycling; soil development; protection of fauna and other flora), but the role most important at the Hanford Site is the ability of plants to minimize recharge by efficiently removing water stored in the near-surface soil. Key plant parameters that control water uptake are when plants are active during the year, the presence of bare spots within the plant community, rooting depth and density, and plant-specific uptake parameters.

Two types of plant communities were considered in the simulations: a deep-rooted shrub community and a shallow-rooted cheatgrass community. Table C.3 shows the parameters used to represent the two communities. Figure C.2 shows the variation in shrub leaf area index. The cheatgrass biomass was specified as 220 g/m^2. All of the plant parameters are identical to those used by Fayer et al. (1999) and Fayer and Szecsody (2004).

Table C.3. Plant Parameters for UNSAT-H Simulations (Source: Fayer et al. 1999)

Parameter Description	Parameter Value	
	Shrub	Cheatgrass
PET partition function	LAI	Cheatgrass
Active period (start day; end day)	60; 334	60; 151
Bare fraction	0.69	0.577
Maximum rooting depth (m)	2.0	0.6
Root density coefficients		
a =	0.217	1.17
b =	0.0267	0.131
c =	0.0109	0.0206
Plant uptake potentials (MPa)		
h_n =	0.003	0.003
h_d =	0.1	0.1
h_w =	7.0[(a)]	2.0[(a)]

LAI= Leaf area index.
PET= Potential evapotranspiration.
(a) Suction heads in the Smoot et al. (1989) backfill simulation were restricted to 10^4 cm (1 MPa), so h_w values for these simulations were reduced to 5000 cm (0.5 MPa).

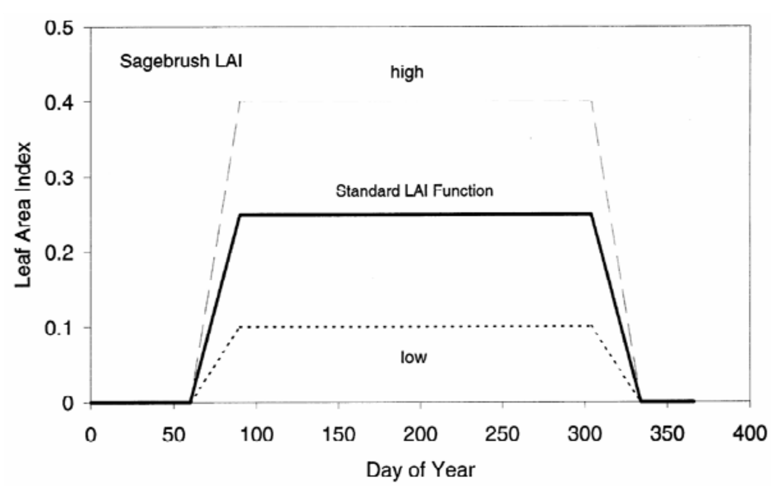

Figure C.2. Leaf Area Index for Sagebrush (after Fayer et al. 1999). Only the standard function was used for this report.

C.3 Results

Each 50-year simulation required less than 0.5 hour of dedicated time to run on a personal computer. In most cases, repeating the weather sequence just once was enough to establish a condition indicative of the long-term average. In several cases, three to four repetitions of the weather sequence were required for the profile to achieve a condition indicative of the long-term average. In some cases, one sequence was enough to establish that the soil profile was drying and that further repetition of the weather sequence would dry out the profile even more.

Table C.4 shows the average long-term deep drainage rate for all simulations conducted. The average rate was calculated for all 50 years of the last simulation sequence. For those simulations that indicated drying, the rate was assigned a value of <0.1 mm/yr. The symbol "<" was used to indicate the uncertainty of specifying such a small rate.

Table C.4. Simulated Long-Term Drainage Rates for Natural Soil Types Using UNSAT-H and Hanford Meteorological Station Weather Data from 1957 Through 2006

Soil Type	Simulated Long-Term Drainage Rates (mm/yr)		
	Shrub	Cheatgrass	No Plants
Rupert sand	1.9	33.0	44.7
Burbank loamy sand	5.2	40.0	53.1
Ephrata sandy loam	0.2	11.7	22.8
Hezel sand	<0.1	6.1	8.7
Esquatzel silt loam	<0.1	6.9	8.6

The results listed in Table C.4 reveal that the general responses to vegetation on all soil types are the same—rates are highest when plants are absent and lowest when plants are deep-rooted shrubs. Cheatgrass reduces drainage, but not nearly as much as the shrub. These results are in accord with previous simulation studies (e.g., Rockhold et al. 1995; Fayer and Szecsody 2004). The simulation period includes some extreme events. Two back-to-back years had annual precipitation amounts that were nearly double the long-term average rate. The period also includes a 24-hr record precipitation of 48.5 mm. This amount is nearly equivalent to the predicted 1,000-yr 24-hr amount of 51.6 mm (Hoitink et al. 2005).

The results in Table C.4 also reveal a distinct difference in the recharge estimates for Rupert, Burbank, and Ephrata soils compared to the Hezel and Esquatzel, regardless of vegetation condition. The key reason for the difference is the presence of the silt loam material in the Hezel and Esquatzel soils. Silt loam has been identified as an excellent candidate material for surface barriers (e.g., Gee et al., 1989; Wing and Gee, 1994). Numerous measurement (Gee et al., 1992; Ward et al. 2005; Fayer and Gee 2006) and computer simulation efforts (Fayer and Szecsody 2004) have confirmed that performance. Neither soil type has been identified as being in or near any of the SST WMAs, but both are present on the Central Plateau.

A third observation regarding the natural soil results in Table C.4 is the lower recharge rates for the Ephrata sandy loam versus that for the Rupert and Burbank soils. This result largely reflects the impact of the small difference in hydraulic properties. Given the spatial variation in Ephrata sandy loam, Burbank loamy sand, and Rupert sand, and the possibility that boundaries between them are not clearly known, we conclude that predictions of site-specific rate estimates will require site-specific information on soil conditions.

Table C.5 shows that the average recharge predicted using the backfill properties from Smoot et al. (1989) was higher than the 130.7 mm/yr predicted by Smoot et al., despite using a higher suction limit of 10^4 cm (which should have increased evaporation and reduced drainage). There are too many differences between the code used then and the one used now to expect precise concurrence between the results. However, a likely contributor to the difference is the Smoot et al. use of surrogate weather data. For example, their predicted annual precipitation after 1988 never exceeded 22.5 cm in a year. The actual record (1989 through 2006) shows two consecutive years—1995 and 1996—when annual precipitation was 31.3 and 31.0 cm, respectively. Such unusual events can be the most significant determinants of recharge.

Table C.5. Simulated Long-Term Drainage Rates for Backfill Using UNSAT-H and HMS Weather Data from 1957 Through 2006

Soil Type	Simulated Long-Term Drainage Rates (mm/yr)		
	Shrub	Cheatgrass	No Plants
Upper Suction Head Limit = 10^4 cm; h_w = 0.5 MPa for Shrub and Cheatgrass			
Smoot backfill	80.7	121.4	143.5
Smoot backfill with modified VG function	48.7	91.1	107.8
S-SX FIR backfill	4.4	19.8	22.2
Upper Suction Head Limit = 10^6 cm; h_w = 7 MPa for Shrub, 2 MPa for Cheatgrass			
Smoot backfill with modified VG function	31.8	75.7	91.0
S-SX FIR backfill	2.3	19.4	22.2

Table C.5 shows that allowing plants to grow on backfill having the Smoot et al. properties reduces drainage but not nearly as much as on the other soil types. Cheatgrass reduced drainage by 15% (relative to the simulation without plants), which is much less than the 25 to 60% reduction seen for other soil types. The reduction was even less for shrubs—on the backfill, shrubs reduced recharge by 44%, while other soil types saw a 10- to 100-fold reduction. The difference in plant effects may be explainable, given the suction head limit placed on the simulations of the Smoot et al. backfill.

With the modified Smoot et al. (1989) backfill properties, the simulation results changed markedly. Drainage rates were lowered by 25 to 40% for all three cases. These results show that recharge predictions are very sensitive to the hydraulic properties of the backfill material, particularly in the drier range of water content.

The drainage predictions made using the S-SX FIR backfill properties were significantly lower (by factors of 6 to more than 10) than when using the Smoot et al. backfill. Compared to the results using the modified Smoot et al. backfill, the drainage reduction was still significant (factors of 4.5 to 10).

The simulations using the modified Smoot et al. backfill and S-SX FIR backfill properties were repeated with a higher suction limit of 10^6 cm and higher wilting point (h_w) values. These changed values are equivalent to the values used in the soil simulations in Table C.4. Table C.5 shows that predicted drainage rate response was different. For the modified Smoot et al. backfill, the drainage rates dropped 17 to 35%. For the S-SX FIR backfill, the drainage rate for the shrub case dropped about 50%, barely changed for the cheatgrass simulation, and did not change at all for the case without plants.

Table C.6 shows the variable impact that the depth of the bottom boundary has on simulated drainage rates. The Smoot backfill showed essentially no effect—drainage was the same whether the boundary was at 1 m or 4 m. As Figure C.1 illustrates, the Smoot backfill exhibits hydraulic properties indicative of very coarse-textured material (e.g., maximum decrease in retention at high water potential). In contrast, the S-SX FIR backfill shows an increasingly larger impact on drainage as the bottom boundary is shifted upward. Compared to the estimate generated using the 4-m boundary, the estimate for the 2-m boundary is only 4% higher. This small difference suggests that tanks buried 2 m below the soil surface may not significantly impact estimates of recharge rates, at least for the isothermal conditions considered in this report. As the depth of the bottom boundary is moved from 2 to 1 m, the impact on estimated recharge increases. With the boundary at 1 m, the estimated recharge is 20% higher than it is when the boundary is at 4 m. The difference in results between the two soil types in Table C.6 shows that soil hydraulic properties play a significant role in determining the sensitivity of recharge to the presence of the lower boundary position in the profile and, by association, the impact of buried objects like tanks.

Table C.6. Simulated Long-Term Drainage Rates for Backfill Using UNSAT-H and HMS Weather Data from 1957 Through 2006 (upper suction head limit = 10^6 cm)

Depth of Bottom Boundary (m)	Simulated Long-Term Drainage Rates (mm/yr)	
	Smoot Backfill with Modified VG Function	S-SX FIR Backfill
1.0	91.0	26.6
1.5	91.0	24.0
2.0	91.0	23.0
4.0	91.0	22.2

C.4 Conclusion

A suite of simulations was used to estimate recharge rates for soil and plant conditions pertinent to the SST WMAs. The soil conditions included five soil types found on the Central Plateau and two backfill types. The simulations were conducted using a 50-year sequence of weather collected at the Hanford Site from 1957 through 2006. This sequence was repeated until the results remained unchanged, to uncouple results from assumed initial conditions.

The simulations of natural soils confirmed what earlier studies have shown—namely, that drainage rates are higher when soil is sandier rather than siltier and when plants are shallow-rooted annuals like cheatgrass rather than deep-rooted perennials like shrubs (Gee 1987; Gee et al. 1992; Fayer and Gee 2006). In all cases, a lack of plants resulted in the highest recharge rates.

The simulations of backfill material without plants confirmed earlier studies that drainage rates could be in excess of 100 mm/yr. The simulations with plants showed how much the drainage rates could be lowered if plants were allowed to establish on the backfill.

The results illustrate clearly that there is a need for hydraulic properties of the backfill material in order to predict recharge in tank farms. To our knowledge, there are no characterization data (physical or hydraulic) for the near-surface backfill material residing directly on any of the SST WMAs.

In addition, a more complete analysis of the impact of buried tanks on recharge ought to consider the multi-dimensional movement of water above a tank as well as the thermal history of the tank.

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