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Contaminant Transport in the Unconfined Aquifer

Input to the Low Level Tank Waste Interim Performance Assessment

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Abstract: This report describes briefly the Hanford sitewide groundwater model and its application to the Low-Level Tank Waste Disposal (LLTWD) interim Performance Assessment (PA). The Well Intercept Factor (WIF) or dilution factor from a given areal flux entering the aquifer released from the LLTWD site are calculated for base case and various sensitivity cases. In conjunction with the calculation for released fluxes through vadose zone transport, the dose at the compliance point can be obtained by a simple multiplication. The relative dose contribution from the upstream sources was also calculated and presented in the appendix for an equal areal flux at the LLTWD site. The results provide input for management decisions on remediation action needed for reduction of the released fluxes from the upstream facilities to the allowed level to meet the required dose criteria.

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CONTAMINANT TRANSPORT IN THE UNCONFINED AQUIFER
Input to the Low Level Tank Waste
Interim Performance Assessment

1.0 INTRODUCTION

This report describes briefly the development of Hanford sitewide model and its application to the Low-Level Tank Waste Disposal (LLTWD) interim Performance Assessment (PA) (Mann et al. 1996).

The waste stored in the disposal facilities will migrate downward through vadose zone and into the unconfined aquifer. The waste entering the unconfined aquifer will be accessible to the public. Therefore, the transport in the unconfined aquifer was evaluated in two different scale models for this performance assessment. A large-scale, sitewide groundwater model was used for evaluating transport of contaminants from the LLTWD site to the Columbia River. A small-scale model (pumping model), that was the primary focus of the PA, was used for evaluating impacts from the transport of contaminants to a hypothetical well 100 m downstream from the LLTWD site. The small-scale model is a subregional model of the sitewide model with a fine grid.

A sitewide groundwater model for Hanford Area has been developed and calibrated for multiple purposes. The development and calibration of the sitewide model are described in detail by Law et al. 1996. The primary objectives to use the sitewide model for this PA are: (1) to develop a conceptual model describing the general flow field for the post-Hanford era because the present regional flow characteristics have been noticeably altered by the Hanford liquid waste discharges into the soil column, (2) to ensure that hydrogeologic parameters used for both the sitewide model and the small-scale disposal facility model are consistent and adequately simulate the site hydrogeologic characteristics, and (3) to evaluate the regional distribution and concentration trends of contaminant plumes emanating from the LLTWD site.

A summary of sitewide model development and calibration is discussed in Sections 2, 3 and 4. An overview of the conceptual model for the Hanford unconfined aquifer is provided in Section 2. Emphasis is placed on the description of the recharge and discharge characteristics of the Hanford Site. In Section 3, the model setup, boundary conditions, and hydrogeologic parameter selections are described. A brief discussion of the model calibration is provided in Section 4. The application of the sitewide model to the LLTWD site for the post-Hanford era is discussed in Section 5. A comparison of the simulated water levels with pre-Hanford data is provided. Such comparison provides additional confidence that flow field characteristics have been correctly simulated. A set-up of small-scale model and its application to the base case is discussed in Section 6. An evaluation of the Well Interception Factor (WIF) or dilution factor is also provided to check the consistency between the large-scale model and small-scale model. Section 7 discusses various sensitivity cases. Section 8 describes an alternative base case at the Grout Facility where the LLTWD site may be located. Section 9 discusses uncertainties inherent in the sitewide model application to the LLTWD site that could influence hydraulic flow regime and contaminant plume distribution estimates. The implications of these uncertainties on the conservatism of the chosen base case model relative to groundwater

contamination is discussed. In addition, to put the Defense Nuclear Facilities Safety Board (DNFSB, 1994) 94-2 recommendation in perspective, a relative dose effect from upstream sources is briefly discussed in Appendix A.

2.0 CONCEPTUAL MODEL AND DATA FOR THE HANFORD UNCONFINED AQUIFER

The geology of the Hanford Site is described in detail in DOE (1988) and Delaney, et al. (1991). Significant geologic features used in the sitewide model are discussed below.

The Hanford Site lies within the Pasco basin in south central Washington State. The major stratigraphic units are, in ascending order, the Columbia River Basalt Group, the Ringold Formation, Pre-Missoula Gravel, and the Hanford Formation.

The Columbia River Basalt Group is composed of a thick sequence (>5,000 m) of basalts that form the bedrock in the Columbia Basin. The Ringold Formation consists of variably indurated clay, altered mud and sand, fine to coarse sand, and multi-lithologic gravel.

The Pre-Missoula Gravels are up to 25 m thick, contain less basalt than the underlying Ringold Gravel and the overlying Hanford deposits, and sharply truncate underlying strata. The pre-Missoula gravels are approximately time equivalent to the early "Palouse" soil and the Pio-Pleistocene alluvium. The gravels are interpreted as mainstream deposits of the Columbia River (Delaney et al. 1991). The Hanford Formation consists of gravel, fine- to coarse-grained sand, and silt.

The unconfined aquifer is contained within the Ringold Formation and the Hanford formation of the Pasco Basin. This basin is bounded by Umtanum Ridge, Yakima Ridge, and the Rattlesnake Hills on the west; Rattlesnake Mountain on the southwest; the Saddle Mountains to the north; and the Palouse Slope on the east. The Columbia River traverses through the basin and forms the northern and eastern boundary of the unconfined aquifer for the Hanford Site. The top of the Columbia River basalts is generally considered to form the bottom of the unconfined aquifer. The top of basalt is based on surface geologic maps of the Pasco Basin (Reidel and Fecht 1994a and 1994b) over 300 boreholes drilled into the basalt sequence.

The sources of natural recharge to the unconfined aquifer are the areas of high relief west of the Hanford Site. From the recharge areas, the ground water flows down-gradient to the discharge areas at the northern and eastern edge of the site, primarily the Columbia River. The Yakima River is another recharge source on the south of Hanford Site. The recharge from Yakima River appears to be keeping contamination from the 200 Areas from moving southward toward the city of Richland.

Two drainage basins, Cold Creek and Dry Creek, receive an annual average precipitation of about 41 cm, compared to that at Hanford of about 16.5 cm. In addition to surface runoff from the drainage basins, there is ground-water recharge from the basins across the western boundary of the site. Water from Cold Creek and Dry Creek infiltrates the stream beds and migrates to the water

table. In addition, there is considerable irrigation in the basins that provides seasonal recharge to the unconfined aquifer system west of the Hanford Site.

There is also a potential for groundwater recharge to the unconfined aquifer along the southwest boundary of the Site; i.e., from runoff along the front of Rattlesnake Mountain. However, the recharge is thought to be small.

Natural areal recharge from the precipitation at the Hanford Site is highly variable, ranging from near zero to more than 10 cm/yr depending on the vegetation and soil texture. Recharge is generally negligible when soils are fine-grained with a vegetative cover (Johnson 1993, Routson et al. 1988, Gee 1987, Rockhold et al. 1990). When soils are coarse-grained and there is no vegetation, recharge is likely during the years of high precipitation. Fayer and Walters (1995) developed a natural recharge map based on the data from the lysimeters, tracer studies, neutron probe measurements, and computer modeling. In that map, two distinct recharge zones are apparent: a slightly high recharge zone ranging from 2 to 5 cm/yr distributed mainly in the southeast of the 200 East Area extending to the Columbia River; a low recharge zone of less than 0.5 cm/yr distributed over most of the other areas of the Hanford Site. However, high recharge ranging from 5 to 10 cm/yr scatters over the 200 West Area, 200 East Area, 100 Area, and in the vicinity of Washington Public Power Supply System (WPPSS) near the Columbia River.

The Columbia River along the northern and eastern boundary of the site is a discharge zone for the unconfined aquifer. This was true under pre-Hanford conditions and remains so today. River fluctuations influence the unconfined aquifer for several hundred feet from the river's edge, but over a year's time these fluctuations cancel out.

3.0 HANFORD GROUNDWATER FLOW AND TRANSPORT MODEL

3.1 MODEL SETUP

A three dimensional, finite element code, VAM3D-CG (Huyakorn and Panday 1994), is used to simulate moisture movement and contaminant transport at the Hanford Site. Testing of the VAM3D-CG computer code, version 3.1, and its previous version 2.4b, was conducted to establish confidence that the code is ready for use in performance assessment applications at the Hanford Site (Lu and Langford 1994 and 1995).

The model grid consists of six layers with 2,474 elements and 2,611 nodes per layer. It has a total of 14,844 elements and a total of 18,277 nodes. A horizontal finite element grid size of 600 m x 600 m was selected with element heights varying depending on the saturated thickness of the aquifer and the high-conductive pre-Missoula/Hanford formation. Vertical discretization was performed using 6 elemental layers, the top three (layers 4, 5, and 6) represent the pre-Missoula/Hanford formation, and the bottom three represent the underlying Ringold formation.

3.2 HYDROGEOLOGIC PROPERTIES

The hydrogeologic properties of the aquifer materials within the system control the magnitude of flow and determine the configuration of the water table. Eighteen hydrogeologic material zones were used to characterize various sediments. The material identification numbers for the upper layers (Layers 4, 5, and 6) and lower layers (Layers 1, 2, and 3) are shown in Figures 1 and 2, respectively. The hydraulic property input data, such as hydraulic conductivities, specific storage, and porosity are taken from Law et. al. (1996) and listed in Table 1.

Table 1. Element Material-Zone Flow Properties.^a

Zone	$K_{xx}(m/y)^b$	$K_{yy}(m/y)$	$K_{zz}(m/y)$	S_s^c	Effective porosity
1	.800E+03	.800E+03	.800E+02	.100E-05	0.10E+00
2	.190E+04	.190E+04	.190E+03	.100E-05	0.25E+00
3	.500E+04	.500E+04	.500E+03	.100E-05	0.10E+00
4	.650E+04	.650E+04	.650E+03	.100E-05	0.10E+00
5	.140E+05	.140E+05	.140E+04	.100E-05	0.25E+00
6	.720E+05	.720E+05	.720E+04	.100E-05	0.25E+00
7	.260E+05	.260E+05	.260E+04	.100E-05	0.10E+00
8	.300E+05	.300E+05	.300E+04	.100E-05	0.25E+00
9	.430E+05	.430E+05	.430E+04	.100E-05	0.25E+00
10	.550E+05	.550E+05	.550E+04	.100E-05	0.25E+00
11	.770E+05	.770E+05	.770E+04	.100E-05	0.25E+00
12	.899E+05	.899E+05	.899E+04	.100E-05	0.25E+00
13	.140E+06	.140E+06	.140E+05	.100E-05	0.25E+00
14	.300E+06	.300E+06	.300E+05	.100E-05	0.25E+00
15	.750E+06	.750E+06	.750E+05	.100E-05	0.25E+00
16	.113E+07	.113E+07	.113E+06	.100E-05	0.25E+00
17	.183E+07	.183E+07	.183E+06	.100E-05	0.25E+00
18	.213E+07	.213E+07	.213E+06	.100E-05	0.25E+00

^a Taken from Law et al. (1996)

^b K_{xx} , K_{yy} , K_{zz} are principal component of hydraulic conductivity tensor

^c S_s is specific storage

Figure 1. Material Zone Map of Upper Layers (4, 5, 6)

Figure 2. Material Zone Map of Lower Layers (1, 2, 3)

3.3 BOUNDARY CONDITIONS

Boundary conditions are applied to the model as conceptualized in previous section. Influx boundary conditions are used along Cold Creek and Dry Creek representing natural recharge from the two drainage basins. Inflow boundaries along Yakima River are prescribed head conditions, with head values supplied from observed water table elevations in the region.

The model treats the Columbia River as a boundary that is held constant in time but variable in distance along the river, with numerical values obtained from five river stage gages (four in 100 Area; one in 300 Area) over one-year period (1992-1993). The river stage elevation values themselves are undocumented. The data used to determine the values were collected according to the protocol described in the report by Campbell (1994), and the transducer equipment and its operation are described in Campbell and Newcomer (1992). The head values are compared with the pre-Hanford water table map which shows river stages interpolated from water table contours. This comparison implies that the average annual river stages are relatively stable over the decades. The Priest Rapids Dam average yearly discharge data supports this assessment. The average yearly discharge is $2.97 \times 10^3 \text{ m}^3/\text{s}$ in 1992, which is in good comparison to the average of $3.34 \times 10^3 \text{ m}^3/\text{s}$ over the period of 1960-1994 (McMahan 1995).

A no-flow boundary condition was used at the bottom of the model that is taken to be the top of the lower mud unit of the Ringold Formation where it exists; otherwise it is the top of basalt. The exclusion of the mud unit in the simulation is appropriate because of the immobility of the contaminant as it sinks into this portion of the aquifer. Also, Gable Mountain, Gable Butte, and a subcrop of the Umtanum Ridge are treated as elements of "zero" hydraulic properties so that there is no flow in those elements.

The model assumes no recharge from precipitation. Most of the Hanford Site is vegetated and the upper soil layers are relatively fine grained. Slightly higher recharge areas are downstream of most waste storage and disposal sites. Although the recharge in the 200 Areas where processing of nuclear fuels has taken place is relatively high, in view of a high volume of the artificial discharges resulting from Hanford operations, the natural recharge is thought to have no or minor effects on the direction of groundwater flow.

4.0 GROUNDWATER MODEL CALIBRATION

A three-phase calibration (Law et al. 1996) was performed. The initial phase was to calibrate to the steady-state period of 1976-1979. During this period, the water table position remained relatively constant. Also the volumes of waste water disposed to ground remained relatively constant during this time.

The second phase started with the steady-state calibrated model of 1979, and simulated in a transient mode the period of 1979 through 1993. The results are examined at 8 and 14 yr (i.e., for 1988 and 1993 conditions).

The final phase is to confirm the hydraulic calibration by simulating the movement of tritium plume.

4.1 STEADY-STATE CALIBRATION (1979)

Groundwater generally flows from regions of recharge to regions of discharge. The general flow pattern is interrupted locally by groundwater mounds and the basalt outcrops and subcrops at the Hanford Site. The groundwater mounds were beneath the high volume disposal sites: U Pond in the 200 West Area, B Pond east of 200 East Area, and Gable Mountain Pond north of 200 East Area. They resulted from artificial recharge from liquid disposal at a rate that exceeded the ability of the sediments to transmit this water (Graham et al. 1981).

The artificial recharge resulting from Hanford operations provides the source term and is input at nodes along the top boundary of the model. The flux boundaries at 1979 were estimated from records reported by Slinger (1980): the discharge at three high volume disposal sites, U Pond, B Pond, and Gable Mountain Pond; the discharge at moderate volume disposal sites, 216-S-10 and 216-S-11 cribs near U pond; and the discharge at moderate volume disposal site, 216-A37-1 crib in 200-East Area. U pond began receiving waste discharges in 1944 and was decommissioned and covered over in 1984. B-pond began receiving waste water in 1945. Two expansion ponds were constructed in 1984 adding 50% more capacity (Zimmerman et al. 1986). Gable Mountain Pond began receiving waste water in 1957. Decommissioning of Gable Mountain Pond began in 1984, but it continued to receive reduced waste water discharges. There are a number of low-volume discharge sites, but they are assumed to have no a direct impact on the unconfined aquifer flow directions.

Several cases were run to determine how best to represent boundary conditions resulting from natural areal recharge on the west and southwest sides of Hanford Site. It was decided to use constant head boundary conditions only for the calibration exercise to quantify the inflow from the recharge area. Constant head boundary conditions are the result of the natural recharge from Cold Creek and Dry Creek and artificial recharge from U Pond discharge conditions. These conditions are, therefore, correct only for the 1979 hydrologic flow conditions.

The simulated and measured water tables representing the 1979 steady-state flow conditions are shown in Figure 3. The calibrated water table agrees favorably with the field measurements (Law et al. 1996).

After calibration, inflow fluxes from the boundary nodes were used to replace those fixed head boundary nodes. The model reproduced the water tables and flow streamlines under 1979 conditions. These influx boundary conditions represent the natural recharge conditions from the two drainage basins under 1979 conditions and are expected to be representative of average conditions over time unless regional precipitation characteristics change drastically. Recharge to the system from Cold Creek is $0.12 \text{ m}^3/\text{s}$ and Dry Creek is $0.51 \text{ m}^3/\text{s}$, which compares to field estimates of 0.028 to $0.28 \text{ m}^3/\text{s}$ inflow. The model simulated discharge of $1.47 \text{ m}^3/\text{s}$ from the unconfined aquifer to the Columbia River (Law et al. 1996).

4.2 TRANSIENT SIMULATION AND COMPARISON (1979-1993)

The calibrated steady-state simulation representing December 1979 conditions was next used as the initial condition for a transient simulation of 14 yr. Nine nodes are assigned for flux boundary conditions representing

Figure 3. Water Table (1979)

ponds and cribs that have been discharging large volumes of water to ground surface. Flux boundary condition over the ponds and cribs were varied annually in a stepwise fashion to represent annual average recharge at the respective locations. The selected facilities represent about 95% of the liquid disposed to the ground (Law et al. 1996). Figure 4 shows the simulated and measured water levels at 8 year, representing 1988 conditions. Figure 5 shows the simulated and measured water levels at 14 year, representing 1993 conditions. These water levels are compared to the observed values, and are in agreement to within the calibration targets (Law et al. 1996).

Figure 6 shows the water table changes that occurred between 1988 and 1979. The mound in the 200 West Area is noted to decline while the mound in the east of 200 East Area is seen to increase. The decreases in 200 West Area is due to the reduced recharge in the area. Meanwhile, the increases in the east of 200 East Area are due to the increased recharge to B pond. Figure 7 shows the water table changes that occurred between 1993 and 1988. The mound declines significantly in the east of 200-East Area due to the decreasing recharge to B pond during this period.

Figures 8 and 9 show the pathlines for 1988 and 1993, respectively, representing various particle travel times and flowpaths. Each mark represents five years' travel time. In both plots (Figures 8 and 9), the particle flowpaths originate in the same locations but some of them end in different locations because of a significant change of the flow field that occurred in the vicinity of B pond. One of the pathlines that moves from west to east and then to southeast for 1988 moves from west to east and then to northeast for 1993. The other (south border of 200 West Area) that moves from west to east and then to southeast for 1988 moves from west to east and then to north for 1993. These shifts in direction are consistent with the change in B Pond recharge during this time frame.

4.3 TRITIUM PLUME CALIBRATION (1979-1993)

The calibration was performed by analyzing historical tritium plumes behavior at the Hanford Site from December 1979 to December 1993. The simulation used the flow field generated by the calibrated flow model over the same time period. Mechanisms that govern the movement of the tritium plume include advection, dispersion and decay.

Since the advection of the tritium plumes is controlled primarily by the calibrated flow field. The longitudinal dispersivity A_L , transverse dispersivity A_T , and vertical dispersivity A_V are additional parameters which regulate the simulated behavior of the tritium plumes. The dispersivity is generally scale dependent and results from the spatial and temporal variations of the flow field. Gelhar et al. (1992) compiled dispersivity values determined from field-tests at 59 different sites including two investigations at the Hanford Site. One investigation examined the 200 East tritium plume using two-dimensional transport model. They came up with dispersivity values of $A_L = 30.5$ m and $A_T = 18.3$ m (Ahlstrom et al. 1977).

The calibration of tritium transport model was initiated by including the observed 1979 tritium distribution with the calibrated steady-state flow model. Transient simulations ran through 1993. Boundary conditions included a zero concentration for inflow at Cold and Dry Creeks and the Yakima River, and zero normal concentration gradient condition at the discharge nodes along

Figure 5. Water Table (1983)

Figure 6. Water Table Level Changes (1979 to 1988)

Figure 7. Water Table Level Changes (1988 to 1993)

Figure 8. Simulated Pore Velocity & Pathlines Map (1988)

Figure 9. Simulated Pore Velocity & Pathlines Map (1993)

the Columbia River. The annual average tritium mass inflow was provided at inflow locations (cribs and ponds) consistent with the flow model. A net of 2.78×10^{11} pCi of tritium is injected during the 14 yr simulation period. The injected tritium mass were much less than the existing total mass in the aquifer (i.e., the observed 1979 tritium plume).

The initial tritium plume (measured in 1979) is presented in Figure 10. The simulation used dispersivity values described above. The dispersivity values were then adjusted to reproduce the tritium plumes measured in 1988 and 1993. The final adjusted longitudinal dispersivity A_L , transverse dispersivity A_T , and vertical dispersivity are 30.5 m, 3 m and 3 m, respectively. The A_L value remains the same as calculated from two-dimensional transport model of 200-East tritium plume study reported in Ahlstrom et al. (1977). But, the A_T value is six time smaller as that obtained by Ahlstrom et al. (1977). These changes in the A_T value is expected because a three-dimensional model is used in this calibration rather than a two-dimensional model. The ratio of A_T/A_L is about 1/10 and is consistent with ratios reported in the literatures (Gelhar et al. 1992).

The simulated results of tritium distribution in 1988 and 1993 are presented in Figures 11 and 12, respectively. The tritium unit were converted to pCi/l to compare with the measured values. The comparison of the simulated and measured tritium plume in 1988 and 1993 are presented in Figures 13 and 14, respectively. For the 14-yr simulation period, the domain lost a net 2.16×10^{14} pCi, with 1.75×10^{13} pCi being discharged across the boundary into the Columbia River, and 1.98×10^{14} pCi having decayed. The tritium plumes compare fairly well with the field data.

5.0 APPLICATION OF THE SITEWIDE MODEL TO THE DISPOSAL SITES

To apply the site-wide groundwater model to the LLTWD site, the post-Hanford flow conditions were projected by assuming that the artificial recharges from U Pond, B Pond, Gable Mountain Pond and other cribs have completely ceased and the mounds underneath these discharge sites have dissipated. The transient state of the dissipating mounds can be ignored because the contaminant flux for conservative radionuclides will take several hundreds of years to reach the unconfined aquifer, given the expected average infiltration rate of <0.5 cm/yr.

The projected Hanford flow conditions were then used as the basis to project a representative contaminant plume emanating from the LLTWD site. Relative downstream concentrations of the contaminant were calculated at different distances from the source and compared with the small scale model result for consistency.

5.1 POST-HANFORD WATER TABLE

Figure 15 shows the result of a steady-state simulation representing post-Hanford water levels. Figure 16 shows the simulated water table changes that occurred between post-Hanford and 1993. The decreases of water levels are about 10 m in 200 West Area, 4 m in 200 East Area, and 6 m in the vicinity of B Pond.

Figure 10. Simulated Tritium Plume (1979)

Figure 11. Simulated Tritium Plume (1988)

Figure 12. Simulated Tritium Plume (1993)

Figure 13. Tritium Plume (1988)

Figure 14. Tritium Plume (1993)

Figure 15. Simulated Water Table (Post-Hanford)

Figure 16. Water Table Level Changes (1993 to Post-Hanford)

Assuming that there will be no significant changes in the regional recharge and discharge characteristics, the post-Hanford water table configuration should be very similar to the pre-Hanford water table configuration. If the assumption is correct, the predicted water table in the post-Hanford era can be verified by water level measurements in pre-Hanford era or during its early days in operation. Because the spatial hydraulic property distribution is time invariant, close comparison would, therefore, provide confidence that selection of the generalized hydraulic property distribution over the site is reasonably accurate on a regional scale.

Zimmerman et al. (1986) examined how liquid discharges to ground have changed the configuration of the water table with time from 1950 through 1980 at the Hanford Site. The hydrographs for selected wells presented in the report started as early as 1950. The measurements at 1950 were used for comparison in this study. Although Hanford operation started at 1944, only wells near U Pond; B Pond; and T Pond, which have significant discharges during the early days in operation, would be affected. The well locations used for comparison are presented in Figure 17 along with the predicted water table contours for post-Hanford. Only Well 399-8-1 (w20) is not on the plot because it is at 300 Area near the Columbia River.

The comparison between the predicted post-Hanford water levels and the measured water levels in 20 selected wells are presented in Table 2. They are in general in a good agreement except for four wells. One of the exceptions is Well 699-25-70 (w9) at which the measurement is lower than the predicted one by 4.6m. However, the predicted water levels and measurements agree very well at the downstream well (w12) and upstream well (w5). The other significant discrepancy is Well 699-55-89 (w2) at which the measured water level is 3.9 m higher. This happened in the area in which the hydraulic gradient was the highest; the water level dropped from 130 m to 124 m in 1.2 km distance. For the other two wells, 699-55-70 (w7) and 699-50-30 (w16), the measured water-levels are 2.7 m and 2.8 m lower than predicted ones, respectively. Both wells are in the vicinity of the basalt outcrops.

On the other hand, the water level measurements in Well 299-45-42 (w14, Figure 17) and Well 299-26-1 (w8, Figure 17) in 1950 were 124.7 m and 119.3, respectively, compared favorably with 124.4 m and 119.2 m (see Table 2) in the post-Hanford model prediction. This gave a very close gradient value over 1.8 km from the east edge of the 200 West Area to the 200 East Area.

These comparisons indicate that the generalized model of the flow field is reasonable. However, the existence of the discrepancies in the comparison at some areas indicates the local heterogeneity of geohydrologic properties. The sitewide model does not have sufficient resolution to accurately predict the local scale behavior of the aquifer. Thus, site specific data are essential for an accurate prediction for a specific site. When site specific data are not sufficient or a long-term prediction is required, conservatism should be adopted in developing conceptual model for the disposal site. In summary, the ability to represent local variability with the regional model is limited by the availability of input data (e.g., the areal density of well locations and amount of hydrologic data) and the adequacy of the numerical discretization that can reasonably be configured to perform the analyses.

Table 2. Comparison of Predicted Post-Hanford Water-Levels with that Measured at 1950.

Well # on the plot	Well	Predicted Post-Hanford Water Level (m above msl)	Water Level Measured in 1950 (m above msl)	Predicted - Measured (m)
w1	699-63-90	122.2	121.9	0.3
w2	699-55-89	125.9	129.8	-3.9
w3	699-49-79	128.3	129.5	-1.2
w4	699-39-79	129.9	129.5	0.4
w5	699-25-80	130.6	130.1	0.5
w6	699-65-72	118.9	120.0	-1.1
w7	699-55-70	125.5	122.8	2.7
w8	699-45-69A	124.4	124.7	-0.3
w9	699-25-70	127.6	123.0	4.6
w10	699-60-60	118.7	120.1	-1.4
w11	299-36-61A	119.7	119.8	-0.1
w12	699-25-55	119.7	119.2	0.5
w13	699-34-51	119.2	119.2	0.0
w14	699-E26-1	119.2	119.3	-0.1
w15	699-24-33	117.9	118.4	-0.5
w16	699-50-30	117.4	114.6	2.8
w17	699-41-23	117.1	116.0	1.1
w18	699-20-20	117.5	116.7	0.8
w19	699-50-80B	124.9	123.1	1.8
w20	399-08-01	105.1	104.2	0.9

msl = mean sea level

5.2 HYDRAULIC PROPERTIES IMMEDIATELY AROUND THE DISPOSAL SITE

Setup of a small-scale model is based on the information provided by the sitewide model in the vicinity of the disposal site. Figure 18 shows the predicted aquifer thickness in the vicinity of 200 East Area. Across the LLTWD site the aquifer thickness appears to be about 22 m. In the upper three element layers, a high-conductivity pre-Missoula sediment (material type #16) with saturated conductivity value of 1.13×10^6 m/yr covers most area of the site. A low-conductivity zone (#12) with saturated conductivity value of 8.99×10^4 m/yr covers the lower left corner of the site. Underlying the high-

Figure 17. Simulated Water Table (Post-Hanford)
& Well Location

Figure 18. Simulated Aquifer Thickness (Post-Hanford)

conductivity pre-Missoula sediment (#16) is the low-conductivity sediment (#7) which has saturated conductivity value of 2.6×10^{-4} m/yr and characterizes the sediment in the Ringold Formation.

While the current water level beneath the site resides in the high-conductivity Lower Gravel Sequence of the Hanford Formation (Reidel et al. 1995), the predicted water table at the post-Hanford era will reside approximately at the boundary of Ringold Formation (about 119.3 m above the mean sea level). In view of the uncertainty inherent in the regional model about where the exact boundaries are between the low- and high-permeability zones, a conservative modeling approach is warranted. Therefore, the base case assumes that water table will reside in Ringold Formation. If the water table would reside in the high-permeability zone, a high volume of water would participate in mixing and result in lower concentration in the downstream wells.

Figure 19 shows the predicted pathlines representing the particle travel time from chosen locations in the 200 Areas to the Columbia River. Pathlines are dominantly from west to east across the LLTWD site. Note that each mark represents 20 year travel time instead of 5 as shown in Figures 8 and 9. Further evaluation of the contaminant transport from the LLTWD site to the river will be discussed in the later section.

6.0 BASE CASE: AREAL SOURCE REPRESENTING THE CONCEPT 1

The LLTWDs was selected to be located west-south-west from the Grout Facility (see Figure 18) at the beginning of this study. An areal source representing the Concept 1 is used for the base case. For Concept 1 (Eiholzer, 1995), six large vaults (38 m long by 19.5 m wide by 9 m high) are placed end to end, creating a structure 246 meter long (extra spacing being concrete walls) and ten rows of vaults are placed together, creating a structure at least 208 meter in width. For this model, the disposal facility is 250 meter in length (the dimension parallel to the aquifer flow) by 200 meter in width (the dimension perpendicular to the aquifer flow).

6.1 SIMULATION SETUP

The primary objective of the following calculations is to determine the well-intercept-factor. The well-intercept-factor (WIF) is defined as the ratio of concentration at the receipt well and the concentration entering the aquifer (i.e., the concentration at the vadose zone).

In order to calculate the WIF at 100-m downstream well, a small-scale model (pumping model) representing a subregion of the sitewide groundwater model underneath the LLTWD site was developed. The model has four element layers with 2,501 nodes and 2,400 elements per layer, for a total of 12,505 nodes and a total of 9,600 elements. The model planar size is 1.5 km by 0.8 km consisting of two material zones 12 and 7 as shown in Figure 20. The saturated hydraulic conductivity k_s values of 8.99×10^{-4} (zone 12) and 2.6×10^{-4} m/yr (zone 7) representing the "E" sequence of the Ringold was used for all

Figure 19. Simulated Pore Velocity & Pathlines Map
(Post-Hanford)

Figure 20. Material Zone Map, Only Zones 7 and 12:
Used for Small-Scale Model

layers. Material type #7 has porosity of 0.1 and the material type #12 has porosity of 0.25. The element size is 25 m by 20 m. The bottom of the model is flat; an average value of basalt top over the area was used.

Values of hydraulic head taken from the sitewide model were used for the fixed head boundary on the west and east of the subregion model. Boundary conditions for transport model include a zero concentration for inflow at the west boundary, and zero normal concentration gradient condition at the east boundary to allow the contaminant across the boundary of the model region.

The recharge rates (infiltration rate) are assumed to be 1 mm/yr for the disposal facility (barrier) and 3 mm/yr for the surrounding area. A uniform solute flux of 1×10^{-3} Ci/yr/m² was assigned over the entire disposal facility. The solute flux is the product of the contaminant concentration in the water and the infiltration rate of the water. For instance, for a given infiltration rate of 0.001 m/yr the concentration contaminant before entering the aquifer is about 1 Ci/m³.

The model simulations were done with a well placed 100 m from the boundary of the disposal facility. A low pumping rate of 10 liters per day (2 liters per person; 5 persons in the family) was assumed.

6.2 SIMULATED RESULTS AT 100-m DOWNSTREAM WELL

The results of the model are presented in Figures 21 through 23. Figure 21 shows the concentration contours at $t=120$ yr at the uppermost element layer (layer 4). Figure 22 shows the concentration contour at $t=120$ yr at the vertical cross section containing the pumping well. Figure 23 shows the concentration history plot representing the contaminant leaving the well as a function of time due to pumping. The concentration reaches a steady state at a time of about 30 years with a maximum value of 1.77×10^{-3} Ci/m³. This will give the well-intercept-factor (WIF) of 1.77×10^{-3} if a recharge rate of 1 mm per year is assumed. Since the source term is solute flux which is the product of the solute concentration and the water flux, the results can be interpreted in different ways. The WIF would be 0.89×10^{-3} if the recharge rate is 0.5 mm/yr and 5.31×10^{-3} if the recharge rate is 3 mm/yr.

An additional well-interception-factor at 900 m downstream (200 East Area boundary) is also calculated. If a recharge rate of 1mm per year was assumed, the well-interception-factor at this location is 1.36×10^{-3} . This calculation also serves the other purpose; i.e., to check the consistency of the results from the small- and large-scale models. A comparison will be made in the next section. The impact on the calculation of numerical dispersion generally grows with the size of the grid. The ratio of the grid size of sitewide model and small-scale model is about 30.

6.3 WELL-INTERCEPT-FACTOR AT DISTANT DOWNSTREAM WELLS

The sitewide model was used to estimate the WIF at distant downstream wells. Figure 24 shows the simulated velocity field, pathlines and travel time from the LLTWD site to the Columbia River, and the hypothetical well locations, designated as h1, h2, ..., and h8 along the streamlines. The 200 East Area boundary falls roughly between the locations h1 and h2.

Figure 21. Simulated Concentration Contours; Base Case

Figure 22. Base Case: Cross-Section View

Figure 23. Simulated Base Case: Pump Rate = 0.01 m³/day

Figure 24. Simulated Pathlines & Hypothetical Well Locations

An equivalent contaminant concentration per unit width of disposal site (width is in the direction of north-south) employed in the small-scale (pumping) model is used in this simulation. The total contaminant flux entering the aquifer, however, is three time larger than that used in the small model because the source width (600 m in grid size) is three time larger than the width of the disposal site. However, the concentration, C_0 , remains to be 1 Ci/m^3 before entering of the aquifer if the recharge rate is 1 mm/yr .

The results of the model simulation at $t=400 \text{ yr}$ are presented in Figure 25. The figure shows the contaminant concentration contours at the element Layer 4.

The concentration history plots for eight wells (see Figure 15 for well locations) are presented in Figures 26 and 27. The average well intercept factor for wells, h1 and h2, is about 1.36×10^{-3} , coincidentally the same value as that obtained from the small-scale model at 200 East Area boundary.

Two conclusions are drawn from the calculations:

- A consistent well-interception-factor estimate is shown at the 200 East Area boundary using the small-scale model and large-scale sitewide model. It appears that the numerical dispersion inherent from the coarse grid of the numerical model is insignificant for this model application.
- The well-intercept-factors are generally diminished as the well location moves away from the source location. The well-intercept-factors are listed in Table 3.

Table 3. Well-Intercept-Factor at Distant Downstream Wells.

Locations*	h1	h2	h3	h4
WIF	1.63×10^{-3}	1.09×10^{-3}	6.91×10^{-4}	5.95×10^{-4}
Locations	h5	h6	h7	h8
WIF	1.58×10^{-4}	9.74×10^{-5}	9.02×10^{-5}	6.93×10^{-5}
Remark	200 East boundary is about halfway between wells h1 and h2			

* See Figure 24 for hypothetical well locations.

7.0 SENSITIVITY ANALYSIS

7.1 SENSITIVITY CASE S1: PUMPING RATE AT 100-m DOWNSTREAM WELL

This sensitivity study examines the effect of varying the pumping rate at the 100-m downstream well on the contaminant plume. The analyses are performed for four pumping rates, $10 \text{ m}^3/\text{days}$, $50 \text{ m}^3/\text{days}$, $100 \text{ m}^3/\text{days}$, and $150 \text{ m}^3/\text{days}$. Conceptually, this would reduce the contaminant at the downstream wells if the well capture zone reaches out beyond the contaminated water zone.

Figure 25. Concentration Contour at t=400 yr

Figure 26. Simulated Concentration History at Downstream Hypothetical Wells

Figure 27. Simulated Concentration History at Downstream Hypothetical Wells

The simulated results are presented in three sets of figures, 28a-d, 29a-d and 30a-d. The pathlines are plotted and presented in Figures 28a, 28b, 28c, and 28d, respectively, to show the horizontal capture zones under four pumping rates. Similar plots in a vertical cross-section containing the location of the pumping well are presented in Figures 29a, 29b, 29c, and 29d. These figures (28a-d and 29a-d) provide insight where the water is withdrawn toward the well by pumping.

Figures 30a, 30b, 30c, and 30d provide the concentration history plots at the 100-m well and at the 200 East Area boundary when the 100-m downstream well is being pumped. Table 4 shows the effect of pumping on the well-intercept-factors at these two locations with various pumping rates. In summary, the results show that the pumping effect becomes significant when the well capture zone matches or exceeds the size of the contaminant plume. This is because more fresh water outside the contaminated zone is pulled toward the well for mixing.

Table 4. Effect of Pumping Rates on the Well-Intercept-Factor.

Pumping Rate	Basic Case 0.01m ³ /day	10m ³ /day	50m ³ /day	100m ³ /day	150m ³ /day
WIF at 100m downstream well	1.77x10 ⁻³	1.68x10 ⁻³	1.33x10 ⁻³	1.01x10 ⁻³	7.91x10 ⁻⁴
WIF at 200 East Boundary	1.36x10 ⁻³	1.24x10 ⁻³	8.40x10 ⁻⁴	4.81x10 ⁻⁴	2.58x10 ⁻⁴

7.2 SENSITIVITY CASE S2: RAISE THE WATER TABLE

The geologic layers used in sitewide model are interpreted from limited measurements and might greatly differ at a specific site. One of the uncertainties comes from the boundary of lithologic units. This sensitivity study examines the effect on the contaminated plume by raising the water table by 5 m. As a result, the uppermost layer will reside in the Hanford Formation and its material property number would be "16" instead of "7" (see Figures 1 and 2). The conductivity at that layer increases by a factor of 37 from 2.6x10⁵ m/yr to 1.13x10⁷ m/yr (see Table 1). For the purpose of comparison, the simulation was performed using the same source term as the base case. Conceptually, the values of WIF would decrease in comparison with the base case because a faster flow field provides more fresh water to dilute the contaminants. The model results are shown in Figures 31 and 32. Figure 31 displays the simulated pore velocity distribution. Two apparent differences from base case (see Figure 28a) are shown: (1) a sharp contrast between the high velocity zone (#16) and low velocity zone (#12); (2) the pathlines which move slightly southward in comparison with that in the base case. Figure 32 shows the simulated solute plume. A 100-m downstream well which is located between contours "B" and "A" recorded a maximum concentration of 2.64x10⁻⁴ Ci/m³. This gives a WIF of 2.64x10⁻⁴, decreased by a factor of 6.7 compared to that in the base case. The values are listed in Table 5 along with base case and the other sensitivity cases for comparison.

Figures 28a-28d. Pumping Rates; 10-150 m³/day

Figures 29a-29d. Pore Velocity Streamlines; Pumping Rates=10-150 m³/day

Figures 30a-30d. Concept 1; Pumping at 100 m Well with Pumping Rate 10 m³/day

Figure 31. Simulated Pore Velocity & Pathlines Map;
Sensitivity Case 2: Raise the Water Table

Figure 32. Simulated Concentration Contour; Sensitivity Case 2:
Raise the Water Table

Table 5. Variation of Well-Intercept-Factor.

Sens. Case	Base Case	S2	S3	S4	S5	S6
WIF at a well 100m-downstream	1.77×10^{-3}	2.64×10^{-4}	2.84×10^{-3}	1.89×10^{-3}	1.24×10^{-3}	1.77×10^{-3}
% of Base Case	100%	15%	140%	102%	70%	100%
Sens. Case		S7	S8	S9	S10	Altern. Base Case
WIF at a well 100m-downstream		1.27×10^{-3}	1.02×10^{-3}	3.60×10^{-4}	1.57×10^{-3}	1.64×10^{-3}
% of Base Case		72%	58%	20%	88%	93%

7.3 SENSITIVITY CASE S3: LOWER THE WATER TABLE

Following the previous study, this sensitivity study examines the effect on the contaminant by lowering the water table by 6 m. The material property remain the same as the base case but thickness of the aquifer decreases. A simulation was performed using the same source term as the base case. The simulated pore velocity distribution and the pathlines remain similar as the base case. Figure 33 shows the simulated solute plume. The values of WIF increases by about 40 percent in comparison with the base case because the aquifer is more shallow and less fresh water is available for mixing.

7.4 SENSITIVITY CASE S4: ROTATE THE FACILITY BY 90 DEGREES

This sensitivity study examines the effect on the contaminant plume by rotating the disposal facility by 90 degrees. Conceptually, if the flow direction across the disposal facility is dominantly from west to east, the rotation would put the longest dimension to be perpendicular to the groundwater flow and result in a reduction of contaminant plume because that the amount of waste per unit width is dropped. However, the small gain (disposal site is nearly square) due to the reduction of the waste per unit width was compensated by more waste being brought upward to the center of the plume due to the nearly diagonal flow across the site. Figure 34 shows the simulated solute plume. The values of WIF are nearly the same in comparison with that of base case (see Table 5).

Figure 33. Simulated Concentration Contour; Sensitivity Case 3:
Lower the Water Table

Figure 34. Simulated Concentration Contour; Sensitivity Case 4:
Rotate the Disposal Facility 90 Degrees

7.5 SENSITIVITY CASE S5: REDUCE RADIONUCLIDE DENSITY IN WASTE FORM

This sensitivity study examines the effect of reducing radionuclide density in waste form by a factor of two and increasing the number of vault rows by a factor of two. Conceptually, if the flow direction across the disposal facility is dominantly from west to east, the change would result in a nearly half of WIF value because of double amount of fresh water used for mixing. Figure 35 shows the simulated solute plume. The values of WIF are reduced by 30 percent in comparison with the base case (see Table 5). The reduction is less because the diagonal water fluxes bring more contaminant upward (width of the disposal site is doubled) and the WIF was calculated only at the center of plume.

7.6 SENSITIVITY CASE S6: REDUCE RECHARGE IN 200 AREAS BY A FACTOR OF TWO

This sensitivity study examines the effect of reducing recharge in the modeling area by a factor of two except the disposal site. The case assumes that the future land use in 200 Areas is heavily industrial. Under this assumption the liquid discharges to the ground would likely be regulated and be kept small. Figure 36 shows the simulated solute plume. The value of WIF at a 100-m downstream well remains nearly the same as the base case (see Table 5). The simulation is insensitive to the change of recharge. It appears that the recharge is too small to alter significantly the Darcy velocity in both direction and magnitude.

7.7 SENSITIVITY CASE S7: SHORTER CAPILLARY BARRIER (CONCEPT 2 LAYOUT)

One of the design alternatives for the disposal facility is a consideration using shorter capillary barriers to fend the infiltration more effectively. The design was designated as concept 2 (Eiholzer 1995). In this design, for every row with waste there is additional row of back-filled soil to drain the diverted water. Furthermore, soil will be placed between canisters and between walls and the canisters to provide a wicking medium to wick moisture. As a result, thirteen contaminant vault rows (three more than concept 1) are needed to accommodate the same number of canisters. There are twelve vault-sized back-filled soil separating them. Model element size, 25m by 20m, remains the same. Each vault row has a rectangular area of 250 m in length by 20 m in width. An uniform solute flux of 1×10^{-3} Ci/yr/m² was assigned over thirteen rows of canister vaults, and a pure recharge water flux of 1×10^{-3} m/yr was assigned over the twelve soil rows. Figure 37 shows the simulated solute plume. The value of WIF is reduced by 30 percent in comparison with the base case (see Table 5) and a similar explanation as Case S5 can be applied.

7.8 IRRIGATED FARMING

Large scale irrigation is practiced in land contiguous to the Hanford Site and it is conceivable that similar agricultural practices could evolve on the Hanford Site over time. Certainly, in the near future, there will be no agricultural practice on site, but the possibility that large scale agriculture may occur at the site in the future can not be eliminated. Three

Figure 35. Simulated Concentration Contour; Sensitivity Case 5:
Double the Size of the Facility

Figure 36. Simulated Concentration Contour; Sensitivity Case 6: Reduce the Recharge in 200 Area

Figure 37. Simulated Concentration Contour; Sensitivity Case 7:
Concept 2 Layout

cases are considered: (1) land is irrigated over the north of 200 Area; (2) land is irrigated over all modeling area except the 200 Area and area between 200 West Area and 200 East Area; and (3) land is irrigated over the west of 200 West Area.

The recharge rate of 10 cm/yr was used to model the "irrigated farming" The recharge estimates were based on the following two references.

Dove et al. (1982) estimated the recharge for the Pasco Basin to be 7.4 cm/yr for current conditions. The current condition accounted for water uses associated with agricultural, irrigation, and pumping practices. The current conditions assumes alfalfa as the crop because of the high moisture content it requires. The 7.4cm/yr was a result of 82% of total water input being lost to evapotranspiration. The recharge is both natural & artificial.

A higher recharge rate of 26 cm/yr was estimated by Gutknecht et al. (1980). The precipitation data was based on interpolating from 58 weather stations; the irrigation requirements for crops such as alfalfa and small grain were from the Soil Conservation Service; two actual evapotranspiration (AET) methods were used as the high and low estimations; and the runoff is zero for the basin.

A recharge rate of 10 cm/yr was considered to be appropriate to average the low, high irrigated farming, irrigated and non-irrigated land. The following three scenarios are simulated.

7.8.1 Sensitivity Case S8: Irrigation Over the North of 200 Area

This sensitivity study examines the effect of irrigation north of the 200 Area. A steady state run for the sitewide model was conducted by imposing additional flux boundary conditions on the upper boundary nodes over the irrigation areas. Figures 38 and 39 show the water table and water table increases due to irrigation, respectively. The water table rise on 200-East Area is about 2m. However, the increase in water flux in this 2-m interval is significant because the interval is in the high permeability zone. The directions of flow field are also changed slightly southeastward in the vicinity of the LLTWD site.

To evaluate the effect of irrigation on a 100-m downstream well, the small-scale model with fine grid is used again. The boundary conditions on the small-scale model was assigned according to the new water table generated by the sitewide model with assumed irrigation fluxes. Figure 40 shows the hydraulic head distribution in the small-scale model. Effect of the irrigation on the flow direction is evident. The water table underneath the site increases about 2 m and this additional water will reside in the high-permeability zone. A higher-volume of water flux would participate in the mixing than that in the base case. Figure 41 shows the concentration distribution. Compared to the base case, the WIF value at the 100-m downstream well dropped to 58 percent of the base case value.

Figure 38. Simulated Water Table; Irrigation North of 200 Area

Figure 39. Irrigation Over the North of 200 Area; Water Table Changes with Irrigation and without Irrigation

Figure 40. Simulated Water Table; Irrigation North of 200 Area

7.8.2 Sensitivity Case S9: Irrigation Except 200 Areas

This sensitivity study examines the effect of irrigation over the modeling area except for the 200 Areas and area between 200 West and 200 East Areas. Flux boundary conditions were imposed on the upper boundary nodes over the irrigation area, and a steady state run was conducted again. Figure 42 shows a significant water table rise compared to the base case. The water level contours indicate a gradient pointing from the LLTWD site toward the gap between Gable Butte and Gable Mountain. Figure 43 shows the simulated pathlines. The flow direction is dominantly northward over the entire 200 East Area.

The boundary conditions of the small-scale model was assigned according to the new water table generated by the sitewide model with assumed irrigation fluxes. Figure 44 shows the simulated hydraulic head distribution. The hydraulic gradient is predominately northward and the water table is substantially higher than that in the base case. The uppermost layer in this model is in the high permeability zone (#16). Figure 45 shows the concentration distribution. The peak concentration or WIF value is about 5 times lower than that in the base case because high volume of water flux participate in mixing with same volume of contaminant fluxes.

7.8.3 Sensitivity Case S10: Irrigation Over the West of 200 West Area

This sensitivity study examines the effect of irrigation west of the 200 West Area. Flux boundary conditions were imposed on the upper boundary nodes over the irrigation area; a steady state run was conducted. Compared to water table with no irrigation (Figure 15), Figure 46 show substantial increases in the water levels underneath the irrigation area and 200 West Area. Again, the water table rise on the 200 East Area is small because of a high permeability zone in the area.

The boundary conditions of the small scale model was assigned according to the new water table generated by the sitewide model. Figure 47 shows the hydraulic head distribution generated by the small-scale model. The water levels are slightly higher than that in the base case. This additional water level will, however, reside in the high-permeability zone. Figure 48 shows the concentration distribution. The peak concentration or WIF value reduces to 88% of the base case value (see Table 5).

7.9 SUMMARY OVER THE SENSITIVITY STUDY

Only one sensitivity case S3 results in a higher WIF value. The sensitivity case S3 has a shallower aquifer, and thus a lesser water is available for mixing. The most reduction in WIF value is the sensitivity case S2. The upper portion of the water table for this case is in the high permeability zone, and thus high water fluxes comes to mix with the contaminants and results in a smaller WIF value. Similarly, for irrigation cases, S8, S9, and S10, the WIF values are dropped according to the degree of portion of the aquifer residing in the high permeability zone.

Figure 41. Simulated Concentration Contours; Irrigation
North of 200 Area

Figure 42. Simulated Water Table; Irrigation Everywhere Except 200 Area

Figure 43. Simulated Pore Velocity & Pathlines Map;
Irrigation Everywhere Except 200 Area

Figure 44. Simulated Water Table; Irrigation Everywhere Except 200 Areas

Figure 45. Simulated Concentration Contours; Irrigation Everywhere Except 200 Area

Figure 46. Simulated Water Table; Irrigation West of 200 West Area

Figure 47. Simulated Water Table; Irrigation West of 200 West Area

Figure 48. Simulated Concentration Contours; Irrigation West of 200 West Area

8.0 ALTERNATE BASE CASE: LLTWDs AT THE GROUT FACILITY

The decision on the location of the LLTWDs has not been made yet. The modeling results of the base case and sensitivity cases discussed in this document are on the assumption that the LLTWDs is west-south-west from the Grout Facility as shown in Figure 18. However, the Grout Facility has been considered for LLTWD site recently. Since two sites are only about one kilometer apart, the geohydrologic conditions are not anticipated to be very different and so are the modeling results. To verify this assumption, a modeling run for an alternate base case is performed. Namely, the small-scale subregion model (pumping model) is transported to the area covering the Grout Facility.

The model geometry remains the same as the previous model: four element layers with 2,501 nodes and 2400 elements per layer; model planar size, 1.5 km by 0.8 km; and model element size, 25 m by 20 m. Only one material zone (#7) with saturated hydraulic conductivity value of 2.6×10^{-6} m/yr was used over the entire modeling area. Values of hydraulic head at the Grout Facility area taken from the sitewide model were used for the fixed head boundary on the west and east of the subregion model.

8.1 SIMULATED RESULTS

Again, a uniform solute flux of 1×10^{-3} Ci/yr/m² was assigned over the entire disposal facility. For a given infiltration rate of 0.001 m/yr, the concentration contaminant before entering the aquifer is about 1 Ci/m³. The results of the model simulation are presented in Figure 49 representing the contaminant concentration contours at the element layer 4 at t=120 yr.

The concentration at a 100-m downstream well is about 1.64×10^{-3} Ci/m³; this gives a well-intercept-factor 1.64×10^3 . Compared to the base case, the WIF dropped down slightly to 93% of base case value (Table 5).

9.0 SITEWIDE MODELING UNCERTAINTIES AND IMPLICATIONS FOR THE PERFORMANCE ASSESSMENT RESULTS

The degree of uncertainty inherent in the model comes from the data availability and assumptions made in developing and applying them. A regional model does not have sufficient resolution to accurately predict local-scale behavior of the aquifer.

In addition to the regional versus local characterization issues, projection of future local conditions is considered to be more uncertain in this part of the Hanford Site because the saturated aquifer declines rapidly from south to north across 200 East Area as the basalt bedrock approaches the surface to the north. The groundwater contamination results calculated in this PA are based on the interpretation of long-term hydrologic conditions underlying the LLTWD site using the sitewide model. Alternative interpretations to the projection of the future local conditions need to be addressed.

Figure 49. Simulated Concentration Contours;
Alternate Base Case

A study to evaluate the potential for transport of nickel radioisotopes from the Hanford 218-E-12B Burial Ground (Rhoads et al. 1994) postulated an unsaturated sediment beneath the burial ground extending to almost entire 200 East Area. The postulation was based on the CFEST two-dimensional simulation under an assumption of a low recharge condition.

The desaturation underneath the 218-E-12B burial ground seems supported by the measurements of the adjacent wells. In 1992, thickness was estimated at 1.2 to 5.9 meters for wells 299-E34-7, 299-E34-3, 299-E26-1, and 299-E27-10 which lie progressively south from the site. In late 1995 thickness appears to be 0.4 to 5.2 meters. Further declines are anticipated in the Post-Hanford era.

However, the desaturation condition should be limited to the vicinity of the site. The extensive unsaturated areas are not, however, supported by either the latest Pacific Northwest National Laboratories (PNNL) report on three-dimensional site-wide modeling (Wurster et al. 1995) or the sitewide model (Law et al. 1996) used in this work. Well 299-E26-1 that is located less than 1 km south to the burial ground recorded water level of 119.3 m at 1950 which is consistent with sitewide model prediction in the post-Hanford era (see Table 2). The aquifer thickness at further south Wells 299-E24-7 and 299-E24-18 were estimated to be more than 25 meters at 1994. The sitewide model predicts the thickness of these wells at 22 meter at the post-Hanford era.

The other difference in this sitewide model from the CFEST model is the assumption of the areal nature recharge. One major assumption used in this model calibration was that natural recharge over the Hanford Site has no impact on the hydrologic flow regime. This assumption was considered reasonable during Hanford operation years because of the high volume of artificial recharges. However, the assumption may not be valid for the post Hanford conditions.

Two sensitivity runs were performed using the assumed natural areal recharge rates of 0.5 cm/yr and 5 cm/yr over the entire Hanford Site. The low recharge case is a close approximation of the natural areal recharge map developed by Fayer and Walters (1995). For the case of the lower recharge rate, 0.5 cm/yr, the water table map and the pathlines are presented in Figures 50 and 51, respectively. The water table map and flow field are very similar to the case with no natural recharge (see Figures 15 and 16). The impact on the flow field appear to be minor for the low recharge case.

For the high recharge case of 5 cm/yr, the water table map and pathlines are presented in Figures 52 and 53, respectively. The pathlines originating in the 200 East Area would be generally northward toward the river. This is consistent with the PNNL studies depicted in the report by Rhoads et al. (1992, 1994) assuming a similar recharge rate. However, the water levels generated by the high recharge case are not supported by pre-Hanford water level data. For instance, the predicted water levels (i.e., aquifer thickness) at the West Area are about 12 m higher than the water levels in the pre-Hanford (1944) conditions. This high recharge case is not justified by current climate and the recharge studies performed in the Hanford Site.

If we consider the potential effects of this flow field on groundwater contamination predictions, we qualitatively predict that low contamination concentrations will result because of high recharge. That is because the

Figure 50. Simulated Water Table (Post-Hanford);
Recharge=0.5 cm/yr

Figure 51. Simulated Pore Velocity & Pathlines Map (Post-Hanford);
Recharge=0.5 cm/yr

Figure 52. Simulated Water Table (Post-Hanford);
Recharge=5 cm/yr

Figure 53. Simulated Pore Velocity & Pathlines Map (Post-Hanford;
Recharge=5 cm/yr

water table will reside in high permeability zone and would result in lower concentration in the downstream wells. The results in irrigation cases, S8, S9, and S10 support this assessment.

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APPENDIX A

RELATIVE DOSE EFFECT FROM UPSTREAM SOURCES

1.0 INTRODUCTION

The Defense Nuclear Facilities Safety Board (DNFSB, 1994) in its 94-2 recommendation has stated that past disposal and other practices that could interact with contamination from the disposal facility must be considered in performance assessment analyses. Implementation of the 94-2 recommendation requires composite analysis for multi sources. The potential release of radionuclides from various waste packages (i.e., container and waste form), transport through the soil column, and contamination of the underlying unconfined aquifer at their own space and time is a complex, non-steady state, three dimensional process that needs a simplification to make modeling possible and efficient.

The two general processes that fundamentally control the radionuclide concentration in the groundwater withdrawn from downstream wells (compliance point or points) are the integrated radionuclide flux from the facilities and the volume of groundwater that mixes with and carries the radionuclides released from the facilities. The radionuclide fluxes from the facilities are controlled by recharge, waste release mechanisms, disposal facility characteristics, and the vadose zone transport. And the volume of mixing water is determined by regional flow characteristics, primarily the hydraulic gradient and the aquifer permeability.

To quantify this process, a simplified modular approach has been adopted to facilitate the analyses. That is: (1) to evaluate and calculate each source term separately in terms of radionuclide flux entering into the unconfined aquifer; and (2) to perform a composite analysis at the unconfined aquifer for multi sources using the Hanford sitewide groundwater model. This study focuses on the second module calculation, a composite analysis at the unconfined aquifer.

This approach is justified as long as each source evaluation leads to a conservative result and as long as the dilution controlling processes are recognized and adequately represented by simplified model.

2.0 OBJECTIVE

The primary objective of this study is to calculate the well intercept factor (WIF) or dilution factor at the LLTWD site from a given areal flux which enters the unconfined aquifer released from an upstream disposal facility. Once the released fluxes through vadose zone transport from disposal facilities are known, the incremental doses at the compliance point can be obtained by a simple multiplication. The results of this study can also be used for management decision on remediation action needed for reduction of the released fluxes from the upstream facilities to the allowed level to meet the required dose criteria.

To achieve the objective, the following three steps are taken: (1) to project the general flow field for the post-Hanford era using the calibrated sitewide model; (2) to identify the upstream disposal sites and other sources that could interact with the LLTWD site; (3) to calculate the well intercept factors at the LLTWD site for each upstream source.

3.0 POST-HANFORD FLOW FIELD PROJECTION

A sitewide groundwater model for Hanford Site has been developed and calibrated for multiple purposes (Law et al. 1996). VAM3D-CG, a three-dimensional, finite element code (Huyakorn and Panday, 1994), is used to simulate moisture movement and contaminant transport at the Hanford Site.

A three-phase calibration (Law et al. 1996) was performed. The initial phase was to calibrate to the steady-state period of 1976-1979. The second phase started with the steady-state calibrated model of 1979, and simulated in a transient mode the period of 1979 through 1993. The results are examined at 8 and 14 yr (i.e., for 1988 and 1993 conditions). The final phase is to confirm the hydraulic calibration by simulating the movement of tritium plume.

The post-Hanford flow conditions were projected by assuming that the artificial recharges from U Pond, B Pond, Gable Mountain Pond and other cribs have completely ceased and the mounds underneath these discharge sites have dissipated (Lu 1996).

The post-Hanford regional flow characteristics are expected to be similar to the pre-Hanford condition as long as regional recharge and discharge characteristics are similar. If the assumption is correct, the predicted water table in the post-Hanford era should compare favorably with water level measurements in pre-Hanford era or during its early days in operation. A favorable comparison was reported by Lu (1996).

Figure A-1 shows the result of a steady-state simulation representing post-Hanford water levels. The selected pathlines, showing potential upstream sources: ERDF site, S, SX, SY, and U tank farms, and the U.S. Ecology site, are presented in Figure A-2. The other unlisted sources such as S cribs surrounding Redox, U cribs south of the U/U03 plants, and others can be assessed from the results of this study.

4.0 ASSESSMENT OF WIF VALUES AT LLTWD SITE FROM UPSTREAM MULTI SOURCES

A constant contaminant flux, 1×10^{-3} Ci/yr/m² over 250m by 600m area, was assigned to the top of the unconfined aquifer-beneath the ERDF site. An identical flux was used for WIF calculation for LLTWD Interim PA (Lu 1996).

The results of the concentration contours at 800 yr are presented in Figure A-3. The calculated well intercept factor at well 100-m downstream from the LLTWD Site is 3.01×10^{-4} . Compared to the WIF value of 1.77×10^{-3} , the base case of LLTWD Interim PA, the dose contributed from the ERDF site is about 17% of dose from the LLTWD site if an equal areal flux enters into the unconfined aquifer.

Figure A-1 Simulated Water Table (Post-Hanford)

Figure A-2 Simulated Pore Velocity & Pathlines Map (Post-Hanford)

Figure A-3 Simulated Plume from Unit Source at ERDF Site

A similar calculation was performed for SX tank farm. Figure A-4 shows the simulated concentration contours at 800 yr. This gave the well intercept factor of 2.08×10^{-4} . Compared to the base case of LLTWD PA, the dose contributed from the SX tank farms is about 12% of dose from the LLTWD site. A similar result is anticipated for S and SY tank farms.

Similar calculations are performed for U tank farm and US Ecology site. Results of the simulation are summarized in Table A-1.

Table A-1. Well-Intercept-Factors for Upstream Areal Flux.

Upstream Sources	Source at LLTWD site (Base Case)	Source at ERDF site	Source at SX Tank Farm	Source at U tank farm	Source at Ecology site
WIF at a 100-m Downstream well from LLTWD site	1.77×10^{-3}	3.01×10^{-4}	2.08×10^{-4}	1.25×10^{-4}	8.91×10^{-5}
% of Base Case	100%	17%	12%	7%	5%

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Figure A-4 Simulated Plume from Unit Source at SX Tank Farm

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