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GEOHYDROLOGIC EVALUATION FOR THE 200 AREA
EFFLUENT TREATMENT FACILITY STATE-APPROVED
LAND DISPOSAL SITE -- ADDENDUM TO
WAC 173-240 ENGINEERING REPORT

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7. Abstract

This document provides a geohydrologic evaluation for the disposal of effluent for the 200 Area Effluent Treatment Facility (ETF). This work forms an addendum to the engineering report that supports the completion of Project C-018H, which comprises the ETF (formerly the 242-A Evaporator/PUREX Condensate Treatment Facility).

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GEOHYDROLOGIC EVALUATION FOR THE 200 AREA
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LAND DISPOSAL SITE--ADDENDUM TO WAC 173-240
ENGINEERING REPORT

1.0 INTRODUCTION

This document provides a geohydrologic evaluation for the disposal of liquid effluent from the 200 Area Effluent Treatment Facility (ETF). This work forms an addendum to the engineering report that supports the completion of the ETF.

The content of this report is required by the Washington State Department of Ecology (Ecology) regulations contained in *Washington Administrative Code* (WAC) 173-240-130, "Engineering Report--Industrial Waste Water Facilities," and supports the development of a State Waste Discharge Permit pursuant to WAC 173-216, "State Waste Discharge Permit Program," for the ETF. This addendum will also satisfy (in part) requirements of *Consent Order No. DE 91NM-177* (Ecology and DOE 1991), the 216 Consent Order.

The purpose of this report is to fulfill the requirements of WAC 173-240-130(2)(p) which calls for a geohydrologic evaluation of waste water discharges proposed through land application. The report has been prepared as an addendum to the comprehensive engineering report (WHC 1993) and should be read in conjunction with that document.

2.0 BACKGROUND

In the past, many untreated liquid effluents at the Hanford Site have been discharged directly to the soil column (Figure 1). The favorable characteristics of the Hanford Site, which include isolation, low precipitation, depth to water table, and soil ion-exchange properties had made this a reasonable disposal method. In March 1987, the U.S. Department of Energy (DOE), Richland Operations Office (RL) issued a document entitled *Plan and Schedule to Discontinue Disposal of Contaminated Liquids into the Soil Column at the Hanford Site* (DOE-RL 1987). This plan contains a strategy for implementing alternative treatment and disposal systems for each of the 33 major waste streams (phase I and II liquid effluent) discharged to the soil column. In accordance with this plan, the 242-A Evaporator process condensate (PC) waste stream must have alternative treatment and disposal systems.

The operation of the 242-A Evaporator is vital to continuing operations at the Hanford Site involving tank waste processing and liquid waste storage. To support 242-A Evaporator operations, RL is in the process of applying for a WAC 173-216 State Waste Discharge Permit.

To provide treatment and disposal systems for the 242-A Evaporator PC, the ETF and State-Approved Land Disposal Site (SALDS), have been established. The SALDS will be located north of the 200 West Area (Figure 2). In 1992, the original scope of the ETF was changed as a result of the decision to permanently shut down the PUREX plant.

Two PUREX waste streams were therefore removed from the scope of the ETF. A State Waste Discharge Permit application, in conjunction with an engineering report pursuant to WAC 173-240, must be approved by Ecology to receive a State Waste Discharge Permit (WAC 173-216) for disposal of the ETF treated liquid effluent.

Pursuant to schedules contained in the 216 Consent Order, RL has agreed to submit a 240 Engineering Report in accordance with WAC 173-240-130. The Engineering Report is structured to address the information requirements listed in subsections of WAC 173-240-130 (2). One of those information requirements (WAC 173-240-130 (2)(p)) stipulates that the following will be performed:

"Where discharge is through land application, including seepage lagoons, irrigation, and sub-surface disposal, a geohydrologic evaluation of such factors as:

- (i) Depth to groundwater and groundwater movement during different times of the year;*
- (ii) Water balance analysis of the proposed discharge area;*
- (iii) Overall effects of the proposed facility upon the groundwater in conjunction with any other land application facilities that may be present."*

The geohydrologic evaluation is presented in this document.

3.0 GEOLOGIC SETTING

This section of the geohydrologic evaluation addendum describes the geologic setting of the Hanford Site and the SALDS. Information presented here has been obtained from Reidel and Thornton (1992), and Delaney et al. (1991).

3.1 REGIONAL SETTING

The Hanford Site (Figure 1) is located in the Pasco Basin of south-central Washington State. The Pasco Basin is one of a number of topographic depressions located within the Columbia Plateau Physiographic Province, a broad basin located between the Cascade Range and the Rocky Mountains. The Columbia Intermontane Province is the product of Miocene continental flood basalt volcanism and regional deformation that occurred over the past 17 million years. The Pasco Basin is bounded on the north by the Saddle Mountains; on the west by Umtanum Ridge, Yakima Ridge, and the Rattlesnake Hills; on the south by Rattlesnake Mountain and the Rattlesnake Hills; and on the east by the Palouse slope.

3.1.1 Physiography

The physiography of the Hanford Site is dominated by low-relief plains in the Central Basin and anticlinal ridges of the Yakima Fold Belt (Figure 3). Surface topography at the Hanford Site is the result of the following (1) uplift of anticlinal ridges, (2) Pleistocene cataclysmic flooding, (3) Holocene eolian activity, and (4) landsliding.

Uplift of the ridges began in the Miocene epoch and continues to the present. Cataclysmic flooding occurred when ice dams in western Montana and northern Idaho were breached, allowing large volumes of water to spill across eastern and central Washington. The last major flood occurred about 13,000 years ago, during the late Pleistocene Epoch. Anastomosing flood channels, giant current ripples, berm mounds, and giant flood bars are among the landforms created by the floods. Since the end of the Pleistocene Epoch, winds have locally reworked the flood sediments, depositing dune sands in the lower elevations and loess (windblown silt) around the margins of the Pasco Basin. Generally, sand dunes have been stabilized by anchoring vegetation, except where they have been reactivated if vegetation is disturbed or absent.

3.1.2 Geology of the Hanford Site

A generalized stratigraphic column showing the suprabasalt sediment is provided in Figure 4. The oldest lithostratigraphic unit of interest is the Columbia River Basalt Group, which consists of an assemblage of continental flood basalts covering an area of more than 160,000 km². These flood basalts were erupted from approximately 6 million to 17 million years ago. Within the Pasco Basin, the Elephant Mountain Member of the Saddle Mountains Basalt is generally the uppermost basalt flows. Sandwiched between many of the flows of the Saddle Mountains Basalt are sedimentary interbeds collectively called the Ellensburg Formation. The Ellensburg Formation includes pyroclastic, fluvial, and lacustrine sediments consisting of muds, sands, and gravels.

Following the cessation of the flood basalt eruptions, there was a period of sedimentary aggradation within the center of the Pasco Basin. This is evidenced by the thick sedimentary sequence of the Ringold Formation. The Ringold Formation was deposited from about 3.9 million to 10.5 million years ago (DOE 1988). This formation is up to 185 m (600 ft) thick in the deepest part of the Cold Creek syncline and pinches out against the anticlinal ridges that bound or are present within the Pasco Basin.

The Ringold Formation consists of semi-indurated clay, silt, pedified mud, fine- to coarse-grained sand, and granule to cobble gravel usually divided into the (1) gravel, sand, and paleosols of the basal unit; (2) clay and silt of the lower unit; (3) gravel of the middle unit; (4) mud and lesser sand of the upper unit; and (5) basaltic detritus of the fanglomerate unit. However, recent studies of the Ringold Formation indicate that it is best described and divided on the basis of sediment facies associations and their distribution. Facies associations in the Ringold Formation (defined on the basis of lithology, petrology, stratification, and pedogenic alteration) include fluvial gravel, fluvial sand, overbank deposits, lacustrine deposits, and basaltic gravel. Lindsey (1991), and Delaney et al. (1991) provide additional details regarding the characteristics of the facies associations (Figure 5).

After Ringold deposition, during the late Pliocene to early Pleistocene, there was a period of regional incision followed by soil development and deposition of windblown sediments (Bjornstad 1984). These deposits are called the Plio-Pleistocene unit and the early 'Palouse' soil, respectively. In the eastern portion of the basin, the Plio-Pleistocene unit and the early 'Palouse' soil are not found and, therefore, either were not deposited or were eroded later by the Columbia River or cataclysmic flooding.

The last 2 million years saw a resumption of fluvial deposition punctuated by periods of erosion and deposition by cataclysmic floods (i.e., Missoula floods). The Hanford formation was deposited during the last ice age by cataclysmic floods. These floods inundated the Pasco Basin a number of times during the Pleistocene, often eroding existing sediments (DOE 1988). As flood waters were dammed up behind Wallula Gap, deposition of both coarse- and fine-grained sediments entrained in the flood waters occurred within the basin. As the flood waters gradually drained, both deposition and erosion occurred. This sequence of events created a complex stratigraphy within the Hanford formation, with lenses of sand and silt surrounded by sand and gravel. For the most part, however, fine-grained sediments are found near the margins of the basin and coarse-grained sediments are found in the central part where the flood currents were stronger. Capping the Hanford formation in many areas is a thin veneer of eolian sands and/or recent fluvial deposits.

As deposition of the basalt flows and sediments was occurring, structural forces were causing the downwarping of the Pasco Basin and the uplift and folding of the surrounding basalt ridges. These structural features generally controlled the locations of the ancestral and present-day rivers and their associated sedimentary deposits.

3.2 GEOLOGY OF THE 200-WEST AREA

The structural features of the bedrock at the SALDS trends roughly east-west, parallel to the major geologic structures bounding the area. The Gable Butte segment of the Umtanum Ridge anticline lies to the north and the Cold Creek syncline and Yakima Ridge anticline lie to the south (Figure 3). As a result, the Ringold Formation and underlying Columbia River Basalt Group gently dip to the south off the Umtanum Ridge anticline into the Cold Creek syncline. Major stratigraphic variations occur in a north-south direction, parallel to the structural dip.

Geologic mapping at the Hanford Site has not identified any faults in the vicinity of the SALDS (Figure 3) (DOE 1988). The closest faults are along the Umtanum Ridge-Gable Mountain structure north of the SALDS.

The stratigraphic section that overlies the Columbia River Basalt Group in the 200-West Area and immediate vicinity is divided, from bottom to top, into (1) the Ringold Formation, (2) the Plio-Pleistocene unit, (3) the early Palouse soil, (4) the Hanford formation and (5) Surficial Holocene sediments. The general stratigraphic relationships are shown in Figure 4.

3.2.1 Ringold Formation

The Ringold Formation in and around the 200 West Area is subdivided into four main stratigraphic sequences. These sequences, originally defined in Lindsey (1991), are similar to those that have been defined historically for the Ringold Formation in the 200 West Area south of Gable Mountain (see Myers et al. 1979; Tallman et al. 1981; Bjornstad 1984; DOE 1988). From bottom to top, the four Ringold sequences are (1) gravel unit A, (2) the lower mud sequence, (3) gravel unit E, and (4) the upper Ringold unit (Figure 5). Unit A corresponds to the lower part of the basal unit of DOE (1988), and unit E corresponds to the middle unit defined by DOE (1988). The lower mud sequence corresponds to the upper basal and lower units defined by DOE (1988). Gravel units B and D (Figure 5) appear to pinch out to the north of 200 West Area. Unit C becomes difficult to distinguish from unit E (see Figure 6).

Gravel units A and E are very similar, consisting predominantly of clast-supported, pebble-to-cobble gravel with a fine- to coarse-grained quartzo-feldspathic sand matrix. Variable but generally small (<5%) amounts of silt are found, as well as localized interbeds of sand and mud usually less than 10 m thick. In the 200 West Area, unit A is 0 to 45 m thick, Unit E is up to 90 m thick, and they are usually separated by the muds of the lower mud sequence. Where the lower mud sequence is absent in the northern part of the 200-West Area, it is difficult if not impossible to differentiate the two units. Both units generally thin to the north on to the Gable Mountain anticline and thicken to the south into the Cold Creek syncline.

The lower mud sequence consists of a lower silty deposit dominated by paleosols and an upper clayey deposit dominated by massive-to-laminated clay and lesser silt. The lower mud sequence thickens to the south into the Cold Creek syncline and pinches out to the north (Figure 5). The paleosols correspond to the upper part of the basal unit, and the massive-to-laminated deposits correspond to the lower Ringold unit as described by DOE (1988).

The upper Ringold unit in the 200 West Area consists of the following: (1) laterally discontinuous sand, (2) silty sand, and (3) sandy silt. The upper Ringold unit is relatively thin in the vicinity of the SALDS (0 to 12 m thick) and pinches out near the border of the 200 West Area.

3.2.2 Plio-Pleistocene Unit and Early 'Palouse' Soil

The Plio-Pleistocene unit and early 'Palouse' soil form a laterally discontinuous deposit that is found throughout much of the 200 West Area and surrounding vicinity. This deposit is absent elsewhere beneath the Hanford Site. The Plio-Pleistocene unit disconformably overlies the Ringold Formation and consists of the following: (1) variably developed pedogenic carbonate-cemented silt, sand, and basaltic gravel and (2) less cemented silt and silty sand. At many locations, the unit consists of several carbonate-rich horizons alternating with carbonate-poor horizons. The carbonate-rich horizons consist of massive to highly fractured pedogenic carbonate, similar to those observed by Machette (1985). Individual layers generally range from 0.5 to 1 m in thickness while massive carbonate rarely exceeds 0.5 m in thickness.

The early 'Palouse' soil is an eolian deposit in the northern part of 200 West Area consisting of the following: (1) brown to dark tan silt, (2) clayey silt, and (3) sandy silt. Calcium carbonate is common in the unit although carbonate cement is not observed. The silts that dominate the unit occasionally contain floating grains of basalt, and muscovite is common in the unit. On natural gamma logs, the unit often can be recognized by a high natural gamma response. In the absence of significant calcium carbonate cement, it is often difficult to differentiate the early Palouse soil from the Plio-Pleistocene unit, and they are thus treated as one unit (see Figure 6).

3.2.3 Hanford Formation

The Hanford formation typically has been divided into a variety of sediment types, facies, or lithologic packages. In the most recent reports dealing with the Hanford formation (Lindsey et al. 1991; Reidel et al. 1992), the following three basic facies are recognized: (1) gravel-dominated, (2) sand-dominated, and (3) silt-dominated. These facies generally correspond to the coarse gravel, laminated sands, and graded rhythmities respectively, as defined in DOE (1988), Baker et al. (1991), and Delaney et al. (1991). The thickest occurrences of the Hanford formation are found in the 200 West Area, where it is up to 65 m (213 ft) thick.

Gravel-dominated strata consist of coarse-grained sand and granule-to-boulder sized gravel that display the following: (1) massive bedding, (2) plane to low-angle bedding, and (3) large-scale cross-bedding in outcrop. A matrix is commonly lacking from the gravel, giving it an open-framework appearance. The sand-dominated facies consists of fine- to coarse-grained sand and granules that display plane lamination and bedding, with less common plane and trough cross-bedding in outcrop. Small pebbles and pebbly interbeds (less than 20 cm [8 in] thick) may be encountered. The silt content of these sands is variable, although where its content is low, an open-framework texture may occur. The silt-dominated facies consists of silt and fine- to coarse-grained sand forming normally graded rhythmities. Plane lamination and ripply cross-lamination is common in outcrop. More detailed discussions of the Hanford formation sediments are found in Baker et al. (1991), Last et al. (1989), Delaney et al. (1991), Lindsey et al. (1991), and Reidel et al. (1992).

3.2.4 Holocene Surficial Deposits

Holocene surficial deposits consisting of silt, sand, and gravel form a thin veneer (less than 5 m [15 ft] thick) across much of the Hanford Site. In the 200-West Area, these deposits consist predominantly of laterally discontinuous sheets of eolian silt and fine sand.

3.3 SITE STRATIGRAPHY

The stratigraphy has been determined from two boreholes, 699-48-77A and 699-48-77B drilled in the vicinity of the SALDS. Borehole 699-48-77B was drilled 4.4 m (14.4 ft) southwest of 699-48-77A to a depth of 18 m (60 ft). No differences in stratigraphy were noted between the two boreholes. Borehole 699-48-77A was drilled to a depth of 139 m (455 ft), using the cable-tool drilling method and core barrel sample recovery, where the

Elephant Mountain Member of the Columbia River Basalt Group was penetrated. The borehole stratigraphy is illustrated in Figure 6. This section describes the lithology penetrated by the borehole and the stratigraphic units encountered. The physical properties of these units were determined from this borehole and analyses of core from nearby Boreholes DH-6, DH-12, and DH-13 (see Reidel and Thornton 1992 for locations). Column leach tests and soil chemistry were determined from samples obtained from both SALDS boreholes (699-48-77A and 699-48-77B). The results are discussed in Section 3.5.

3.3.1 Hanford Formation

The Hanford formation is 7 m (23 ft) thick at the borehole site and consists predominately of open-framework gravels with a sandy matrix typical of deposits of the gravel-dominated facies. A thin silt lens occurs at the base of the gravel sequence above the Plio-Pleistocene and early 'Palouse' interval. The Hanford formation thickens both to the north and south of the site.

3.3.2 Plio-Pleistocene Unit and Early 'Palouse' Soil

The combined thickness of the Plio-Pleistocene unit and early 'Palouse' soil is 13 m (42 ft), over a depth range from 7 m (23 ft) to 19.8 m (65 ft). The upper portion probably correlates to the early 'Palouse' soil but, it proved difficult to separate the early 'Palouse' soil from the Plio-Pleistocene unit on the visual inspection of drilling samples alone. The combined Plio-Pleistocene and early 'Palouse' soil consists of interfingering carbonate-cemented silt, sand and gravel, and carbonate-poor silt and sand. Thin pedogenic calcium carbonate lenses occur throughout the unit.

3.3.3 Ringold Formation

The Ringold Formation is 119 m (390) thick in borehole 699-48-77A and is dominated by the gravel sequences. The Ringold Formation begins at 19.8 m (65 ft) depth and continues to the top of the Columbia River Basalt Group at 138.7 m (455 ft). A major cataclysmic flood channelway that is incised into the Ringold Formation occurs approximately 1 km (0.7 mi) north of the site. As a result of significant flood erosion in this channelway, the Plio-Pleistocene and early 'Palouse' interval, the upper Ringold unit, and a large part of Ringold unit E have been removed. The channelway is filled with approximately 45 m (150 ft) of gravel-dominated deposits of the Hanford formation. The exact position of the erosional channelway edge is not known because no boreholes penetrate deep enough in the area between borehole 699-51-75 (located within the channelway) and the SALDS to encounter the base of the channelway (Figure 7). The channel appears to trend west to east, but this has not been confirmed by other drilling.

The sandy sequence (Figure 6) that occurs between the base of the Plio-Pleistocene and early Palouse interval (19.8 m [65 ft] depth) and the top of the uppermost Ringold gravels (unit E) (25.3 m [83 ft] depth) at the SALDS is interpreted to be erosional remnants of the upper Ringold unit. These strata pinch out to the north, west, and east of the site (Lindsey 1991). The upper unit extends to the south into the northern part of the 200 West Area.

The upper unit grades quickly down into the gravels of unit E at 25 to 26.5 m (83 to 87 ft) depth (see Figure 6). These gravels extends to a depth of 93 m (305 ft) with silt content increasing below 51.8 m (170 ft). The next 12.2 m (40 ft) consist largely of sand with a significant gravel and silt component. Studies of analogous outcrops suggest that these deposits are characterized by gravels interbedded in sands. These deposits are underlain by 12.2 m (40 ft) of sandy gravels that in turn overlie 17.3 m (57 ft) of gravels containing only minor sands. It is not clear if these gravels and those found throughout the Ringold Formation have an open framework texture, although such deposits are rare in outcrops of typical Ringold gravels. The base of the Ringold Formation is marked by a thin sandy unit (1 m [3 ft] thick) overlying the basalt at a depth of 138.5 m (454.5 ft).

The lower mud sequence (Figure 5) is not present in borehole 699-48-77A and is interpreted to pinch out to the south (Lindsey 1991). With the lower mud sequence absent, gravel unit E directly overlies gravel unit A, and it is not possible to differentiate the two units. However, if the top of unit A is projected towards the site from borehole 299-W6-1 (see cross-section Figure 7), it would occur near the top of the cleaner gravels at 120 m (395 ft) depth (see Figure 6). Consequently, much of the lower 18 m (60 ft) of the Ringold section at the site may be part of gravel unit A.

3.4 STRATIGRAPHIC CONTEXT

The stratigraphic units penetrated in borehole 699-48-77A are similar to that encountered elsewhere in 200 West Area. A cross section through the site is shown in Figure 7. This section demonstrates that the stratigraphy in borehole 699-48-77A is consistent with the stratigraphy of the 200 West Area.

Gravels of unit E dominate the Ringold Formation in the area of the SALDS and are the principal aquifer. The thickness of the gravels at the SALDS is slightly less than in the nearest borehole to the south (299-W6-1). The Ringold Formation thins to the north beyond the site. The reduction in thickness is principally because of erosion during the Pleistocene catastrophic flooding, although some thinning also resulted from growth of the Umtanum-Gable Mountain anticlinal structure.

When compared with Ringold gravels from boreholes DH-6, DH-12, and DH-13, the grain-size distribution for Ringold gravels in borehole 699-48-77A is skewed to smaller size fragments (pebble size mean = 47%). A likely reason is that the samples from borehole 699-48-77A were obtained from 'hardtool' and core-barrel drilling, which physically broke up the larger rock fragments as drilling progressed, and incorporated them into the samples. This process resulted in producing artificially smaller fragments that shifted the grain-size distribution to smaller sizes.

Continuous core-barrel sampling through the Plio-Pleistocene unit at the SALDS is devoid of any significant calcrete zone. For the most part, the caliche layers are typically thin (<0.5 cm) stringers. Analyses for calcium carbonate (CaCO_3) obtained from borehole 699-48-77A, and from core in boreholes DH-6, DH-12, and DH-13 suggest that CaCO_3 content is less than 10% by weight.

The Hanford formation shows little lithologic variation in the area of the preferred SALDS (Figure 7). It is similar in thickness at borehole 299-W6-1 to the south (Figure 2). The only significant stratigraphic variation observed in the area is the lateral extent of the

Ringold upper unit and the Plio-Pleistocene unit. The Plio-Pleistocene unit is thicker than in Borehole 299-W6-1, and the upper Ringold is slightly thinner. Both pinch out north of the site.

3.5 GEOPHYSICAL LOGGING

Boreholes 699-48-77A and 699-48-77B were both geophysically logged using the WHC Radionuclide Logging System (RLS) spectral gamma-ray borehole survey tool. Logs were obtained for the following: (1) gross gamma, (2) potassium (^{40}K), (3) thorium (Th), and (4) uranium (U). Data are presented in Appendix C of Reidel and Thornton (1992). The survey detected no synthetic radionuclides. This was confirmed by surveys taken of individual samples from the boreholes. The logs were not useful for geologic interpretation because lithologic variations cannot be identified in a spectral gamma-ray survey of a sealed and completed well.

3.6 GEOCHEMICAL CHARACTERISTICS

This section summarizes the results of chemical analysis and leach testing of sediment samples collected from boreholes 699-48-77A and 699-48-77B. Full details of the analyses, including methods and protocols, are presented in Reidel and Thornton (1992).

3.6.1 Sediment Chemistry

The sediment samples collected for analysis are listed in Table 1. These samples were analyzed for the following: (1) metals, (2) anions, (3) radionuclides, (4) volatile organic constituents (VOC), and (5) semivolatile organics. Splits of the sediment samples were also submitted for analysis of hexavalent chromium (Hanford Environmental Information System (HEIS) Numbers B01RH0 through B01RH8).

Analytical results for selected potential contaminants detected in the sediment samples are presented in Table 2 and summarized below.

- Metals—All analytes are considered to be below levels normally associated with contamination. The *Model Toxics Control Act* (WAC 173-340), for example, stipulates the following Method A soil cleanup standards: (1) 20 $\mu\text{g/g}$ for arsenic, (2) 2 $\mu\text{g/g}$ for cadmium, (3) 100 $\mu\text{g/g}$ for chromium, (4) 250 $\mu\text{g/g}$ for lead, and (5) 1.0 $\mu\text{g/g}$ for mercury. The metals content of the sediments are at or below these concentrations.
- Anions and pH—The concentration of anions was below the limits of detection for all samples, except for the following: (1) a sulfate concentration of 37 $\mu\text{g/g}$ in sample B01RG2, and (2) a fluoride concentration of 5 $\mu\text{g/g}$ in sample B01RG8. The pH of all the sediment samples was 9 to 9.4, which is consistent with the observation that the sediments contain moderate amounts of calcareous material.

- Radiochemistry--Activity measurements of the samples indicated maximum gross alpha of 27.4 pCi/g in sample B01RG1, and maximum gross beta of 21.3 pCi/g in sample B01RG2. Gamma activities were reported as ^{137}Cs , $^{234\text{m}}\text{Pa}$, and ^{234}Th and were at or below the limits of detection. The samples were determined to be essentially nonradioactive when measured for total activity, i.e., <200 pCi/g beta-gamma, and <50 pCi/g alpha. Geophysical spectra logging of the borehole indicates that no anthropogenic radionuclides were encountered (Section 3.5).
- Volatile organic constituents--All VOCs were below detection limits except for sample B01RG3 that contained the following: (1) 2-butanone (methylethyl ketone) at 27 $\mu\text{g/kg}$ and (2) 4-methyl-2-pentanone (methyl isobutyl ketone or hexone) at 130 $\mu\text{g/kg}$. These could be lab contaminants. Guidance provided by the *Model Toxics Control Act* (WAC 173-340) suggests that these constituents should not be of concern at these levels.
- Semivolatile organics--All semivolatile target compounds were at or near detection limits with only trace concentrations of benzoic acid and phthalates reported for several samples. Tentatively identified compounds included carboxylic acids. The carboxylic acids could be natural constituents of the sediments.
- Pesticides/Polybicarbonates--All constituents of this group were at or below the limits of detection.
- Hexavalent chromium--This constituent was below the limits of detection in all soil samples (<0.040 $\mu\text{g/g}$ for B01RH0, B01RH1, and B01RH2 and <0.49 $\mu\text{g/g}$ for B01RH3 through B01RH7).

Analyses of the sediment samples did not detect any significant degree of contamination. Hexone was observed in one sample, but at a low level. Trace amounts of fluoride were detected in the sample taken at the water table.

All anions were at the limits of detection except for sulfate, which was present at low levels in the zone that contained pedogenic calcium carbonate.

3.6.2 Column Leach Tests

Sediment split spoon samples collected from borehole 699-48-77A were leached with deionized water to evaluate sediment/water interaction processes and potential mobilization of sediment constituents. This information is evaluated in this section with respect to the potential effect on groundwater quality at the SALDS arising from disposal of water to the vadose zone.

3.6.2.1 Sample Description and Testing Approach. Eight sediment samples were chosen for the leach tests. These samples were leached in two separate test runs of four samples each. One equipment blank and two deionized water blanks were also performed. Table 3 summarizes the tests performed.

Core samples were leached in these tests at a rate of about one-pore-volume per day. The first leachate sample of each test represents about the first 0.6-pore volume, while succeeding samples were collected at about one-pore-volume intervals.

3.6.2.2 Sediment Leachate Chemistry. Concentrations of major cations and anions and pH measurements of the leachate samples are presented in Table 4. Nitrite and hexavalent chromium concentrations were below the limits of detection. No significant concentrations of metals were observed in the leachate samples.

The data presented in Table 4 indicate that the leachate samples had low total-dissolved-solids content and pH values slightly above neutral. Calcium was the predominant cation. Chloride and nitrate were the most abundant anions measured (carbonate was not analyzed). The leachate pH and concentration of calcium is attributed to the calcium carbonate content of the sediments. A relatively high nitrate content was observed in the first leachate sample obtained from Test 2-1185 (42.7 mg/L), but no values above the primary drinking water standard of 45 mg/L were reported.

Analysis of sediment leachates revealed no significant levels of contaminants that might pose a risk to groundwater quality at the site. The calcium content and slightly alkaline nature of the leachates is attributed to the calcium carbonate content of the sediments.

This conclusion presupposes that the hydrochemical properties (especially the pH) of the waste water will be comparable with the deionized leach water used in the tests.

4.0 HYDROGEOLOGIC CONDITIONS

This section of the geohydrologic evaluation addendum describes the hydrogeologic conditions at the SALDS. Information presented here has been obtained from Reidel and Thornton (1992), Koegler and Reidel (1992), and Laurenz (1993).

4.1 REGIONAL SETTING

The multiaquifer system within the Pasco Basin consists of the following four primary units: (1) suprabasalt sediment of the Hanford and Ringold Formations, (2) Saddle Mountains Basalt, (3) Wanapum Basalt, and (4) Grande Ronde Basalt. Groundwater occurs both within the suprabasalt aquifer system, consisting of fluvial and lacustrine sediments, and within the system of deeper confined to semiconfined aquifers in the basalt flow tops, flow bottom zones, and sedimentary interbeds within the three basalt units (DOE 1988). These deeper aquifers are intercalated with aquitards consisting of dense basalt flow interiors. Vertical flow and leakage across the aquitards are inferred from water level or potentiometric surface data, but the flow and leakage are not quantified and direct measurements are not available (DOE 1988).

4.1.1 Suprabasalt Aquifer System

The suprabasalt aquifer is the uppermost regionally extensive aquifer beneath the Hanford Site. It lies at depths ranging from less than 0.3 m (less than 1 ft) near West Lake and the Columbia and Yakima Rivers, to less than 100 m (less than 350 ft) near the center of the Hanford Site. Groundwater within the suprabasalt aquifer system is contained within the glaciofluvial sands and gravels of the Hanford formation and the fluvial/lacustrine sediments of the Ringold Formation. The position of the water table beneath the western portion of the Hanford Site is generally within the Ringold gravel unit A and E. In the northern and eastern portions of the Hanford Site, the water table is generally within the Hanford formation. Hydraulic conductivities for the Hanford formation range from 610 to 3,050 m/d (2,000 to 10,000 ft/day), which are much greater than those of the silty gravel units of the Ringold Formation (Law et al. 1986). Hydraulic conductivities for gravel unit E of the Ringold Formation range from 0.2 to 61 m/d (0.6 to 200 ft/d), (Lust et al. 1989). The main body of this aquifer occurs within the silty gravel units of the Ringold Formation.

The base of the uppermost aquifer system is defined as the surface of the uppermost basalt flow. On a local scale, the silts and clays of the fine-grained basal units of the Ringold Formation form a confining layer for the overlying coarse facies of the lowermost portion of the Ringold. The suprabasalt aquifer system is bounded laterally by anticlinal basalt ridges and is approximately 150 m (500 ft) thick near the center of the basin.

Sources of natural recharge to the suprabasalt aquifer are as follows: (1) rainfall and runoff from the higher bordering elevations, (2) water infiltrating from small ephemeral streams, and (3) river water along influent reaches of the Yakima and Columbia Rivers. The movement of precipitation through the unsaturated (vadose) zone has been studied at several locations on the Hanford Site. The results are summarized by Gee et al. (1992). Conclusions from these studies vary, depending on the location studied. Some investigators conclude that no downward percolation of precipitation occurs on the 200 Areas Plateau where the sediments are layered and vary in texture, and that all moisture penetrating the soil is removed by evapotranspiration. Others have suggested downward water movement below the root zone from tests conducted near the 300 Area, where soils are coarse-textured and precipitation was above normal. Artificial recharge occurs from the disposal of large volumes of waste water on the Hanford Site (principally in the 200 Areas), and from large irrigation projects surrounding the Hanford Site.

Groundwater flows through the aquifer system from recharge areas to the west, and discharges to areas primarily located along the Columbia River. This general west-to-east flow pattern is interrupted locally by groundwater mounds in the 200 Areas. From the 200 Areas, there is also a component of groundwater flow to the north, between Gable Mountain and Gable Butte. Figure 8 illustrates the water table configuration beneath the Hanford Site for June 1990.

Water levels in the suprabasalt aquifer have risen in the past because of artificial recharge mechanisms such as excessive application of imported irrigation water or the impoundment of surface streams. Waste water ponds on the Hanford Site have artificially recharged the suprabasalt aquifer below the 200 East and 200 West Areas. The increase in water table elevations was most rapid from 1950 to 1960 and apparently had

nearly reached equilibrium between 1970 and 1980 when only small increases in water table elevations occurred. Waste water discharges from the 200 West Area were significantly reduced in 1984, and the water levels there are now slowly declining.

Waste waters discharged on the Hanford Site have reached the suprabasalt aquifer. The primary constituents that have reached the aquifer are tritium, ^{129}I , ^{106}Ru , ^{99}T , uranium, nitrate, and chromium (DOE 1987). The groundwater is routinely and extensively monitored to track the movement of contaminants and to determine if any impacts exist from the Hanford Site to the public. Groundwater monitoring reports are produced annually (e.g., Johnson 1993).

4.1.2 Basalt Confined Aquifer System

Confined aquifers are present in the sedimentary interbeds of the Ellensburg Formation, as well as in interflow zones that occur between dense basalt flows of the Saddle Mountains Basalt, Wanapum Basalt, and Grande Ronde Basalt. The main water-bearing portions of the interflow zones are networks of interconnecting vesicles and fractures of the flow tops and flow bottoms.

Local recharge to the shallow basalts is believed to result from infiltration of precipitation and runoff along the margins of the Pasco Basin. Regional recharge of the deep basalts is thought to result from interbasin groundwater movement originating northeast and northwest of the Pasco Basin in areas where the Wanapum and Grande Ronde Basalts crop out extensively (DOE 1988). Groundwater discharge from the shallow basalt is probably to the overlying aquifers and to the Columbia River. The discharge area(s) for the deep groundwaters is presently uncertain, but flow is believed to be generally southeastward with discharge speculated to be south of the Hanford Site (DOE 1988).

Erosional "windows" through the dense basalt flow interiors provide the potential for direct interconnections between the suprabasalt and uppermost confined aquifers. Graham et al. (1984) indicated that some contamination was present in the uppermost confined aquifer (Rattlesnake Ridge) south and east of Gable Mountain Pond. Graham et al. (1984) evaluated the hydrologic relationships between the Rattlesnake Ridge aquifer and the suprabasalt aquifer in this area and delineated a potential area of inter-communication beneath the northeast portion of the 200 East Area, where significant mounding of the water table resulted from infiltration of B-Pond discharge. This is likely to have provided the driving head to push contaminated water deeper into the aquifer sequence in the 200 East Area.

4.2 SITE HYDROGEOLOGY

Information about the hydrogeologic conditions at the SALDS is based primarily on data from borehole 699-48-77A as well as other boreholes drilled in the vicinity (Figure 9). Groundwater was encountered at about 66.6 m (218.72 ft) depth, approximately 139 m (456 ft) above mean sea level. The water table at the site is located within the Ringold Formation, giving a saturated aquifer thickness of about 72 m (236 ft) above the underlying basalt. Borehole 699-48-77A was completed as a RCRA groundwater

monitoring well with a 4-inch diameter, 10-slot, 20-ft stainless steel wellscreen installed between 64.7 and 70.8 m (215.28 and 235.58 ft) below ground level.

The well development work was performed by pumping the well with a 1.1 kW (1.5 hp) Grundfos submersible pump, and included a small-scale pumping test. The small-scale pumping test was performed on May 18, 1992, and involved pumping for 5 hours at a rate of 23.7 L/min (6.25 gpm), for a total discharge of 7,097 L (1,875 gallons). The drawdown at the end of the small-scale test was 4.3 m (14.0 ft). Water-level recovery was relatively rapid (less than 100 minutes) although the water level recovered only to within 0.6 m (2 ft) of the pretest static water level before it stabilized. Instantaneous slug injection and withdrawal tests were performed on May 19, 1992, after water levels had recovered following the small-scale pumping test. Raw data from these tests are included in Appendix E of Reidel and Thornton (1992).

Water levels during development testing were measured using a downhole transducer. Fluctuations in baseline water levels appear to reflect barometric pressure variations but this has not been verified by analysis.

4.3 SITE GROUNDWATER CHEMISTRY

On completion of the borehole and well development, groundwater from borehole 699-48-77A was sampled on June 9, 1992. This sampling is the first of four quarters of groundwater sampling that will provide baseline data prior to construction and operation of the Soil Column Disposal facility (Harris and Delaney 1991). Groundwater quality in the vicinity of the SALDS has been previously reported by Harris and Delaney (1991). The results from this sampling are reported in Tables 5 and 6.

Table 5 is a listing of contamination indicator parameters used to evaluate groundwater analysis from the borehole. Table 6 reports the following field and laboratory data: (1) conductivity, (2) pH, (3) total organic carbon (TOC), and (4) total organic halogen (TOX). Table 6 also reports those data with at least one value above detection limits for the analytical method used. All constituents were below detection limits except the following: (1) coliform, (2) nitrate, (3) carbon tetrachloride, (4) unfiltered iron, and (5) unfiltered manganese. Filtered iron and filtered manganese are both below the detection limits. This suggests that the unfiltered values are an artifact of well construction. Nitrate is below the WAC 173-200 limit. Coliform was detected at 65 coliforms per deciliter (C/dL) and is above the WAC 173-200 value of 1 C/dL. Carbon tetrachloride was detected at 1.3 $\mu\text{g/L}$, which is below the EPA standard of 5 $\mu\text{g/L}$ but above the WAC 173-200 value of 0.3 $\mu\text{g/L}$. The value for carbon tetrachloride is not quantifiable, however, because the contractually required quantitation limit is 5 $\mu\text{g/L}$, i.e., higher than the result reported. This well will be sampled quarterly as part of the Hanford Groundwater Monitoring Program.

4.4 SITE WATER BALANCE

The water balance at the SALDS will be dominated by the disposal of effluent. The maximum delivery rate for treated effluent flowing to the disposal facility is expected to be 568 L/min (150 gal/min). This may be augmented by blowdown at 83 L/min (22 gal/min), giving a combined maximum discharge of 651 L/min (172 gal/min). An

infiltration rate of 148.7 m/yr (10 gal/day/ft²) has been assumed over an area of 2,000 m² (21,600 ft²) for modeling purposes (Lu et al. 1992). Natural recharge was conservatively assumed by Lu et al. (1992) to be on the order of 50 mm/yr (2 in/yr). Results from the many different studies of natural precipitation recharge at the Hanford Site fall in the range from zero to 100 mm/yr (4 in/yr).

Effluent, plus any additional water from precipitation, snowmelt, and runoff, is expected to percolate downwards through the vadose zone and eventually reach the water table. Mounding of groundwater is expected to occur as described in Section 6.2.2. All infiltrated effluent will eventually move offsite laterally under the influence of either the radial hydraulic gradients induced by mounding, or the natural hydraulic gradient in the water table aquifer. There will be no overall change in storage long-term. As a result, the net water balance will be zero (i.e., total inflow will equal total outflow).

5.0 GROUNDWATER MODELING

Effluent treated in the ETF, including the 242-A Evaporator process condensate, contains tritium. The tritium replaces hydrogen in some of the water molecules and has a half life of 12.3 years. Currently, there is no economically feasible method for removing tritium from water. A strategy was developed for disposing of the tritiated water to the ground and letting the slow groundwater migration to the Columbia River provide the time necessary for most of the tritium to decay. The Columbia River is believed to be the first location that the tritium could be encountered by potential receptors. As part of the SALDS site-selection process, groundwater modeling was performed to determine which of the possible disposal areas being considered would be likely to have the longest groundwater travel path to the Columbia River.

The fate of tritiated water from the proposed SALDS has been examined using numerical models that simulate the infiltration and movement of groundwater and dissolved solutes. Two modeling projects are directly relevant: the vadose-zone modeling by Lu et al. (1992) and the plume migration modeling by WHC (1992b). The model domains overlap at the water table beneath the SALDS and allow a comparison to be made for the assessment of groundwater impacts.

The models and their assumptions are summarized in the first two sections of this chapter. The results of the models in terms of predicted groundwater impacts are summarized in Section 6.0.

5.1 VADOSE-ZONE MODEL

Flow and solute transport in the vadose zone at the SALDS site has been simulated by Lu et al. (1992) in Appendix D of Reidel and Thornton (1992). The disposal of tritiated water at the ground surface through a 32-m-wide (105-ft-wide) crib was simulated for an assumed operating life of 125 years. Lu et al. (1992) assumed that the facility would receive effluent at 568 L/min (150 gal/min) with an expected average tritium concentration of 16.2 $\mu\text{Ci/L}$. (Note that both the expected discharge rate and tritium concentration have subsequently been revised as discussed in Section 5.3). An infiltration

rate of 148.7 m/yr (10 gal/day/ft²) is assumed over an area of 2,000 m² (21,600 ft²). No allowance is made for sorption of tritium in the vadose zone or below the water table, but a half-life of 12.3 years is used for radioactive decay of tritium.

5.1.1 Model Configuration

Lu et al. (1992) used a discretized numerical model to simulate solute transport mechanisms in the vadose zone and up to 50 m (165 ft) beneath the water table. The computer code used to run the model was VAM3D-CG (Huyakorn and Panday 1991), a three-dimensional finite element code that simulates water flow and solute transport in saturated-unsaturated porous media. Lu et al. simulated a two-dimensional vertical cross section from the ground surface to a total depth of 120 m (395 ft) and did not use any of the three-dimensional capabilities of the VAM3D-CG code.

The planar cross-sectional model implies a unit width (perpendicular to the plane of the cross section) and is valid for planar flow domains, such as might occur beneath the central portion of a disposal crib that is significantly longer than it is wide. The model does not simulate the three-dimensional flow domains that will likely occur at either end of the rectangular disposal crib. The disposal of effluent was simulated by applying recharge over a width of 32 m (105 ft). The lateral extent of the vadose zone model was restricted to 50 m (165 ft) in the upgradient direction and 100 m in the downgradient direction, giving an overall lateral dimension of 182 m (600 ft) for the modeled cross section. The finite element mesh and assignment of material types for the model domain are presented in Figure 10.

5.1.2 Material Types and Properties

The geologic profile revealed by borehole 669-48-77A was represented in the model by six different material types. Material properties were based on laboratory analyses of six representative samples obtained during the drilling of boreholes 669-48-77A and 669-48-77B, as listed in Table 7. Moisture retention (or drainage) characteristics were developed for each of the six samples by laboratory measurements of matric potential at different moisture contents and fitting a moisture retention curve using the RETention Curve (RETIC) computer code (van Genuchten et al. 1991). The RETIC code embodies the van Genuchten-Mualem model for moisture retention (Mualem 1976 and van Genuchten 1978). The RETIC code was then used to develop the corresponding characteristic curves for unsaturated hydraulic conductivity as a function of moisture retention and laboratory-measured saturated hydraulic conductivity.

5.1.3 Initial and Boundary Conditions

Initial conditions in the vadose zone are assumed to be controlled by steady-state infiltration of natural recharge at the surface at a rate of 0.05 m/yr. This is probably a conservatively high assumption, given the low, but sporadic, nature of rainfall at the Hanford Site. Initial pressure heads in the vadose zone are defined by assuming that the natural (predisposal) infiltration rate equals the unsaturated hydraulic conductivity, thus allowing the (predisposal) moisture content configuration to be defined.

The saturated part of the model was defined by applying a hydrostatic pressure distribution along boundary nodes beneath a depth of 70 m (230 ft). A lateral hydraulic gradient of 2.75 m/km (14.5 ft/mi) was defined for the saturated zone of the aquifer by applying slightly higher pressures on the upgradient side of the model domain (see Figure 10).

5.1.4 Input Parameters

The following geohydrologic parameters were assumed for the unconfined aquifer:

PARAMETER	VALUE
Aquifer storativity	0.2
Longitudinal dispersivity	1.0 m (3.3 ft)
Transverse dispersivity	0.1 m (0.33 ft)
Molecular diffusion coefficient	$1.4 \times 10^{-6} \text{ cm}^2/\text{s}$
Sorption coefficient (tritium)	0.0

The assumed storativity may be conservative, given a total porosity of nearly 40% and volumetric field capacity of 0.259 (see Table 7). The dispersivity values are stated by Lu et. al. (1992) to be conservative in that they allow for less mixing in the vadose zone and the unconfined aquifer. The molecular diffusion coefficient is of the same order of magnitude as values measured by Hajek (1968) for the Hanford formation and the Ringold Formation. The zero sorption coefficient is conservative.

5.1.5 Results

The results of the vadose zone modeling are summarized in Section 6.0.

5.2 PLUME MIGRATION MODEL

The second modeling effort relevant to the SALDS was performed in 1991 (WHC 1992b). It assumes tritium loading at the water table and simulates lateral plume migration within the aquifer until ultimate discharge to the Columbia River.

5.2.1 Background

A two-dimensional numerical model for groundwater flow and conservative solute transport was developed for the Hanford Site representing the uppermost aquifer in the Pasco Basin west of the Columbia River. Computer simulations were conducted to estimate the effects resulting from disposal of tritium-contaminated effluent to three candidate soil column disposal sites. The prime objective of the study was to simulate the movement of tritium plume in groundwater and provide estimates of the tritium concentration in groundwater that would discharge to the Columbia River. A secondary objective was to provide an estimate of the extent of groundwater mounding beneath the disposal site.

The model used was based on several previous modeling efforts. The original groundwater flow model for the site was prepared to support an investigation of alternative soil column disposal locations for process waste streams from the 200 Areas (WHC 1992a). In this original study, the conceptual model for the site was defined, a numerical grid was developed, and the groundwater flow model was calibrated to observed water levels.

The groundwater flow model developed in the initial study was later used to provide estimates of groundwater travel times from eight candidate tritium soil column disposal sites. Two of the eight original sites and one additional site were subsequently modeled in a study in which both travel time and groundwater mounding estimates were produced (WHC 1992b). To better simulate mounding at these soil column disposal sites, additional discretization was added to the numerical grid in the vicinity of the sources.

The latest modeling effort involved a refined analysis of groundwater mounding and an analysis of tritium plume migration from the three sites evaluated in the last of the aforementioned studies. The entire northern portion of the numerical grid was rediscritized at a finer scale to improve solute transport simulation. Effluent disposal and the subsequent migration of tritium plumes from the three candidate soil column disposal sites to the Columbia River were then simulated, groundwater mounding in the vicinity of the disposal areas was estimated, and tritium concentrations at the river were predicted.

5.2.2 Hanford Site Groundwater Flow Model Development

The two-dimensional groundwater flow field was developed using the AFPM (Aquifer Flow in Porous Media) code. This code is part of the Golder Groundwater Package (GGWP), a collection of finite element computer programs for simulation of groundwater flow and solute transport. Solute transport simulations were carried out using the SOLTR (Solute Transport) code, which is also part of the GGWP. Descriptions of the conceptual model, calibration procedures, assumptions, and limitations are included in Appendix A of WHC (1992b). This information is summarized briefly below.

The groundwater flow model for the region was developed and calibrated in the original modeling effort. During this initial study, the simulated region was defined and boundary conditions were specified. Fixed head conditions were applied along the Columbia and Yakima Rivers, no flow boundaries were used where basalt bedrock rises

above the water table and along the base of the Aquifer, and nonzero fixed flux conditions were applied along Dry Creek and Cold Creek valleys. A map of the simulated region is shown in Figure 11.

The modeled area was divided into 27 subregions, each of which was assigned a value for hydraulic conductivity and other material properties. The model was then calibrated to contour maps of water level observations from two separate dates: 1944 and 1979. Hydraulic conductivities and inflows from Dry Creek and Cold Creek valleys were adjusted in the model until reasonably close results were obtained for both the 1944 and 1979 calibration data sets. The original finite element grid for the model contained 976 nodes and 920 elements.

Additional grid discretization was added later (WHC 1992a) resulting in a grid with 1,086 nodes and 1,083 elements. Most of the elements in these two grids were squares with side lengths of 998 m (3,275 ft). The entire northern portion of the finite element grid was then rediscritized at a much finer scale to improve solute transport simulation. The revised grid, contained 4,639 elements and 4,773 nodes. Each square element in the northern portion of the grid had sides 333 m (1,092 ft) in length. The level of discretization used in this study was chosen to better simulate concentrations at the source and to reduce numerical oscillations and numerical dispersion in the solute transport simulations.

The same material properties and boundary conditions used in the original flow model were transferred to the new finite element grid with the exception that the basalt subcrops were simulated in the newest grid by assigning them extremely low hydraulic conductivities. The 1944 and 1979 calibration runs were then simulated using the new grid. Hydraulic head contours simulated using the new grid were visually compared to those generated during the previous studies for both 1944 and 1979, and material properties were slightly changed to more closely match the contours. The hydraulic head contours for the two calibration runs for the new grid are shown in Figures 12 and 13.

5.2.3 Hydraulic Input Parameters and Assumptions

Having established that the model reproduced flow results from previous modeling efforts, it was necessary to specify input parameters for simulating the candidate soil column disposal scenarios. The boundary conditions used for the 1979 calibration run were also used for these scenarios.

A continuous effluent flux rate at the disposal sites of 568 L/min (150 gal/min) was used. To more accurately estimate the extent of the resulting groundwater mound, the area of aquifer over which this flux would be distributed was evaluated by estimating the lateral spreading of the effluent as it seeps downward from the ground surface to the water table.

The dimensions of the sources used in the model were determined as illustrated in Figure 14. The proposed disposal facilities were expected to have dimensions of approximately 67 x 30 m (220 x 100 ft) at the ground surface. For simplicity, the source at the ground surface was represented by a circle of the same area 2,000 m² (22,000 ft²). It was then assumed that as the effluent seeps through the unsaturated zone, it spreads with an angle from the vertical of 20°. This value is consistent with data from a tank leak

at the Hanford Site reported by Smoot et al. (1989). The size of the source at the water table depends upon the distance from the ground surface to the water table, as illustrated in Figure 14. This distance was estimated based on ground surface and water table elevations at each disposal site. After migrating through the unsaturated zone, the effluent flux is assumed to be evenly distributed at the water table over an area that is larger than 2,000 m² (22,000 ft²). The resulting lateral spread of nearly 25 m is very close to the value of 20 m simulated by the modeling of Lu et al. (1992).

Values used at the preferred SALDS (Site 1) were as follows (see Figure 14 for symbols):

MODEL PARAMETERS	VALUES
Ground Elevation	207 m (680 ft)
Water Table Elevation	140 m (460 ft)
Unsaturated Zone Thickness (h)	67 m (220 ft)
Radius of Source at Water Table (rw)	50 m (165 ft)
Area of Source at Water Table	7,850 m ² (84,500 ft ²)
Area of Source in Model	8,350 m ² (90,000 ft ²)

In developing the source dimensions, the possible presence of low permeability horizons, such as caliche layers, was not considered. If a laterally continuous, low permeability horizon is present, a perched groundwater mound could develop at a shallower depth than a mound formed on the top of the aquifer. Lateral spreading beyond the amount assumed would increase the size of the source at the water table, and reduce the initial groundwater concentrations.

Also, the occurrence of vertical gradients within the aquifer are not simulated in this two-dimensional model which inherently simulates a uniform (hydrostatic) head distribution over the full depth of the aquifer. Thus, the true shape of the water table mound is not represented in this model.

The effect of B-Pond was also included in the model (see Figure 13). The assumed flux from the 216-B-3 Pond System (B-Pond) was 62.5 ML/d (16.5 million gal/day). This flux was distributed evenly over a square element with a side length of 998 m (3,275 ft). Future changes in disposal practices at B-Pond, trenches in the vicinity of U-Pond, and the 200 Areas as a whole were not considered in this modeling work.

5.2.4 Solute Transport Input Parameters

Information about the hydraulic head distribution generated by the groundwater flow model is required for input to the solute transport model, which thus determines how solute is advected in the aquifer. Solute transport modeling also requires additional data including the following: (1) dispersivities, (2) retardation factors, (3) effective porosities, and (4) source terms. These input parameters of WHC (1992b) are summarized below.

Dispersivities were obtained for the model from a combination of Hanford Site data and data from other sites with similar alluvial and glaciofluvial sediments. The appropriate dispersivity values to be used are those that are valid for the approximate distance from the source, to the point at which the plume is to be studied. At locations closer to the source, the size of the plume will tend to be overestimated, and at locations farther from the source the size of the plume will tend to be underestimated. In this application, the modeling objective was to estimate concentrations at the Columbia River resulting from tritium plumes emanating from the vicinity of the 200 Areas. Therefore, the length scale of interest was the travel distance from the candidate disposal sites to the river. The smallest of these distances was chosen, so as to conservatively produce tighter plumes with higher concentrations at the river. This corresponds to a longitudinal dispersivity of about 36.5 m (120 ft) and a transverse dispersivity of about 12.2 m (40 ft).

The values used for dispersivity differ from those chosen by Lu et al. (1992). Dispersivity is scale-dependent (Gelhar et al. 1992) and values in modeling must be consistent with the scale or size of the flow domain being modeled.

Effective porosity values, which are required for solute transport simulations, were also used in the original flow modeling to estimate groundwater travel times. These values were assumed to be correlated with hydraulic conductivity, and were estimated to range from 0.15 to 0.25 based upon the migration of existing plumes at the Hanford Site (WHC 1992a). Because the solute of interest in this application was tritium, retardation was ignored by assuming a retardation factor equal to 1.0. A first-order decay rate for tritium of 0.056 years^{-1} (corresponding to a half-life of 12.3 years) was also used. The boundary of the modeled region for the solute transport simulations was treated as a discharge boundary.

The concentration of tritium in the effluent was assumed to be $21 \text{ } \mu\text{Ci/L}$. With a volumetric flux of 568 L/min (150 gpm), the corresponding mass flux of tritium is 17.2 Ci/d . This mass flux was distributed evenly over the source elements representing the water table of the SALDS.

5.2.5 Interpretation of Two-Dimensional Concentrations

In a two-dimensional model, the solute mass is spread instantaneously over the entire thickness of the aquifer. In reality, a plume resulting from effluent disposal at the surface is likely (at least initially) to be confined to the uppermost portion of the aquifer, although it will eventually disperse vertically downward to some extent. Vertical anisotropy in the aquifer will limit this vertical mixing.

Because the two-dimensional model spreads the simulated plume over the entire thickness of the aquifer, concentrations in the aquifer will be proportional to the thickness of the aquifer at the source. This problem can be approximately corrected by scaling the simulated concentrations by a factor to reflect that the plume is not vertically mixed over the entire aquifer. This factor is the ratio of the aquifer thickness at the source to an estimate of the actual mixing depth for the plume over the scale of interest.

For this application, rather than estimating a plume mixing depth, a depth of 6 m (20 ft) was assumed. This corresponds to the typical depth over which a monitoring well is screened in the aquifer. If the actual mixing depth is less than 6 m (20 ft), this approximation provides an accurate estimate of the concentration that would be measured in the well because the plume will mix over the entire length of the screened interval as the well is being pumped for a sample. If the actual mixing depth is greater than 6 m (20 ft), the predicted concentrations will be conservative (overestimated) because concentrations decrease as mixing depth increases. The aquifer thickness at the SALDS is 57 m (187 ft) and the corresponding correction factor by which two-dimensional concentration results were multiplied was 9.4.

5.2.6 Limitations of the Groundwater Models

Any groundwater modeling effort requires the modeler to make a number of assumptions. An understanding of the assumptions and their rationale is needed to assess the uncertainty of the results. The assumptions and limitations incorporated into the groundwater flow and contaminant transport models are discussed in detail in Appendix A of WHC 1992b. Several of the most important limitations are reiterated below. Quantification of the uncertainty in the modeling results presented here is discussed in Section 5.3.

- The fundamental flow equations used by the AFPM code are derived from standard assumptions for two-dimensional groundwater flow modeling, including the following: (1) no vertical flow, (2) an isotropic medium, (3) vertical averaging of hydraulic conductivity, and (4) deterministic approximation of the flow parameters. This limits the ability of the code to address vertical variability, such as mound heights and depth-variable tritium concentrations.
- The scale dependency of dispersivity is difficult to address in solute transport modeling. Variable dispersivity approaches require extensive effort to implement and were not required in this study to predict concentrations at the Columbia River. This study was based on constant dispersivity values representative of transport over distances of tens of thousands of feet; concentrations predicted closer to the source (on scales of hundreds or thousands of feet) must be considered approximate.
- It was necessary to make some simple, conservative assumptions to represent the size and strength of the solute source at the water table. Also the size and flux of the source at the water table influence the predicted mound height, with smaller sources and greater fluxes resulting in higher mounds. Uncertainties in the size of the source could therefore cause the actual mound height to be either higher or lower than the predicted value.
- Numerical dispersion and numerical oscillations that arise in solving the governing equation for solute transport inherently limit

the accuracy of a numerical solution, particularly near the source where the concentration gradients are high.

- The model neglected the interaction between groundwater and the Columbia River. Transient changes in river elevation are likely to create variable concentrations of tritium at the river over time, with an overall reduction in expected tritium concentrations of water discharging to the river.
- The flow model developed using the AFPM code was calibrated by comparing predicted and actual hydraulic head contours, as discussed in WHC 1992a. The SOLTR code, however, was used to develop a model for predicting the migration of contaminant plumes at locations where no plumes presently exist. The transport model was based upon available Hanford Site transport and flow data and is considered technically correct. Although there is no information for calibrating the model against existing plumes along the flow paths studied, satisfactory results were obtained when the model was used as part of another study to project the movement of existing plumes following other flowpaths on the Hanford Site (WHC 1991).

5.2.7 Model Results

Contours of hydraulic head for the Hanford Site resulting from the operation of the SALDS are shown in Figure 15. Detailed head contours for the estimated extent of groundwater mounding are also shown. These figures show that the mounds influence the water table elevations by 1 foot or more for distances of up to 15,000 feet from the source locations; however, as discussed below, significant changes in groundwater flow directions resulting from the mounds are much more localized in the immediate vicinity of the source.

The estimated height of the mound at the SALDS is 1.5 m (5 ft). The model is not designed to represent the mound realistically in any detail. Groundwater mounding is likely better simulated in the vadose zone modeling (Section 5.1) and is not discussed here.

Disposal at the SALDS was predicted to have relatively little influence on local direction of groundwater movement. Gradient reversals were predicted in the immediate vicinity of each of the mounds, extending approximately 150 m (500 ft) from the source at the SALDS. The lateral distances to points where the groundwater flow directions were essentially unchanged (remaining within about 20 degrees of the original flow directions) were more highly varied, and extended about 300 m (1,000 ft) from the SALDS.

Solute transport simulations were run until a steady state condition was reached (that is, the contours did not change with time). Lines of equal concentration were contoured at times ranging from 5 years up to steady state (see Figures 16 through 20). A smoothing routine based on weighted averaging of the tritium concentrations at

adjacent nodal points was employed to eliminate some of the irregularities attributable to the high concentration gradients near the source.

Noticeable numerical effects were found upgradient of the sources stemming from the numerical oscillation problems. The actual edges of the plumes in these areas could be identified in the original, unsmoothed model output as locations where concentrations became negative, and served as a basis for entering corrected, hand-drawn contours upgradient of the sources. Because of the use of constant dispersivity values representative of the distance from the disposal sites to the river, contours closer than several thousand feet from the source are considered approximate. The numerical effects near the source were not entirely corrected in designing the study because the precise geometry of the plume in the disposal area was not an objective.

Contoured results for the SALDS are shown for 5, 55, 85, 105, and 205 years in Figures 16 through 20. The plume is at steady state at 205 years, and close to steady state at 105 years. As can be seen, the plume migrates relatively slowly at first (because of low hydraulic conductivities assigned in the model), and accelerates as it passes through the gap west of Gable Mountain. Profiles of tritium concentration in groundwater along the edge of the Columbia River were developed for each site correlated to the river distances shown in Figure 21. This modeling work did not include existing tritium plumes in the 200 Area, which are shown in Figure 22. Figures 23 through 26 show the profiles for the plume emanating from the SALDS. Tritium concentrations along the river from that plume were predicted to be less than the regulatory groundwater criterion of 0.02 $\mu\text{Ci/L}$ (20,000 pCi/L) at steady state.

5.2.8 Uncertainty of Modeling Results

The numerical effects experienced near the plume sources accelerated solute transport in the vicinity of the sources during approximately the first few years of the model simulations, artificially reducing the travel distance to the Columbia River by an estimated 5,000 feet. This resulted in a decrease in the travel time and more conservative, higher estimates of tritium concentrations at the river.

Treatment of the basalt subcrops as low permeability units had minor effects on the distribution of tritium in the model. The plumes traveled around the subcrops as expected, following the principal directions of groundwater movement. Because the subcrops were assigned a permeability which was eight orders of magnitude less than the aquifer material, the amount of tritium advected into the subcrops was negligible. A minor amount of tritium was simulated to have entered the basalt because of numerical dispersion and oscillation near the source. These effects are only apparent in the immediate vicinity of the source and are negligible elsewhere.

The tritium plumes from the three candidate sites were modeled using the best available technology and data. In view of the model's successful treatment of actual Hanford plumes (WHC 1991), the predicted tritium concentrations at the river are expected to be reasonably accurate. Note that the simulated concentrations at a *specific location* along the river may be off by orders of magnitude, since a small change in the direction of plume migration could have large effects on concentrations, particularly at the edge of the plume. However, the simulated maximum concentrations discharging into the river are considerably more accurate. The degree of accuracy in modeling results

can be estimated by considering the major sources of uncertainty in the input data. The major potential sources of uncertainty with respect to solute transport modeling are as follows: (1) source representation, (2) dispersivity values, (3) groundwater flow directions and fluxes, and (4) mixing depth.

Because the river is a considerable distance from the sources, the concentrations at the river will be relatively insensitive to the source representation (assuming the flux input is accurate), unless the spatial distribution of the source is grossly misrepresented (such that the advective flow field transports the mass in a significantly different direction). There does not appear to be a gross misrepresentation of the source. Likewise, steady-state concentrations at the river are controlled primarily by the advective flow field, and are not very sensitive to dispersivity. Uncertainty in the dispersivity values would translate to smaller uncertainty in the width of the plume. An estimated uncertainty of less than a factor of two for the tritium concentrations at the river is therefore reasonable.

The two major sources of uncertainty that can affect simulated concentrations are the groundwater flow direction and flux, and the mixing depth. As pointed out above, differences in the groundwater flow direction can have significant impacts on the simulated concentration at a specific point. However, unless the direction of groundwater flow is drastically misrepresented, the effect on the maximum simulated steady-state concentration at the Columbia River would be small. On the other hand, the simulated concentrations are directly proportional to the flux of groundwater moving through the source area (the higher the flux, the more the effluent is diluted). Because the flow model was calibrated based on observed heads, available permeability data, and recharge estimates, and because existing plumes have been reasonably well matched, it is expected that the groundwater flux at the sources is accurate to within a factor of two or three.

The assumed mixing depth of 6 m (20 ft) has a linearly inverse effect on simulated concentrations (the greater the depth, the lower the concentration). A mixing depth of 6 m (20 ft) is considered to be on the conservative side. The mixing depth is limited by the thickness of the aquifer, which is on the order of approximately 30 m (100 ft). Consequently, it is considered that the mixing depth is likely to be accurate to within a factor of five or less. Considering the uncertainties discussed above, it is estimated that simulated maximum concentrations discharging into the river are accurate to within an order of magnitude.

Another aspect of potential model uncertainty relates to the possible existence of high-permeability channels in the sediments at the Hanford Site (USGS 1987). Erosional channels that have been cut into the Ringold and subsequently infilled with more permeable sands and gravels of the Hanford formation can potentially provide preferential pathways for the more rapid movement of groundwater. Such a channel may exist to the north of the SALDS (see Section 3.3.3). If such features are affecting the overall groundwater flow system at the Hanford Site, the impacts will be integrated within such data as the water-table maps, and will be included in any modeling by virtue of the calibration process. However, localized effects could be significant and these may be lost within the inherent averaging that takes place as a model is developed.

The vadose zone and plume migration models used different expected tritium levels in the liquid effluent, consistent with estimates available when the individual modeling efforts were conducted. A tritium level of 21 $\mu\text{Ci/L}$ is used by WHC (1992b); Lu et al. (1992) used 16.2 $\mu\text{Ci/L}$. The effluent characteristics anticipated by WHC (1993) include tritium at 60 $\mu\text{Ci/L}$, between 3 and 4 times higher than the concentrations used for the modeling work. However, this level includes the PUREX waste stream, which will not be included. Without the PUREX waste streams, the tritium level in the ETF effluent is estimated to be 6.3 $\mu\text{Ci/L}$ based on the upper bound of the 90% confidence interval for currently available effluent composition estimates (WHC, 1989), or 30 to 40% of the modeled levels. For a geochemically conservative material such as tritium, activity levels within the flow domain will be directly proportional to activity levels in the disposal effluent. The model results from WHC (1992b) and those from Lu et al. (1992) should therefore be factored down to give the current best estimate of probable radionuclide levels in the groundwater.

Applying a factor of 30% to the plume migration results from WHC (1992b), the predicted maximum tritium levels in groundwater discharging to the Columbia River could be up to 0.0042 $\mu\text{Ci/L}$. This would be well within the regulatory water quality criterion of 0.02 $\mu\text{Ci/L}$ (20,000 pCi/L) for groundwaters of the State of Washington (WAC 173-200-040). Furthermore, the plume migration modeling includes some conservative assumptions which may also affect the estimated tritium levels in discharge to the Columbia River. In particular, it was arbitrarily assumed that vertical dispersivity of tritiated water within the aquifer would be limited to the upper 20 ft. The basis for selecting 20 ft relates to a typical wellscreen length for groundwater monitoring wells; it does not necessarily reflect hydrogeologic factors that could produce this effect, including anisotropy resulting from the dominant horizontal stratification of alluvial sediments. As described in Section 5.2.5, the tritium levels produced by the model were uniformly multiplied by a factor of 9.4 in an attempt to convey the expected restriction in vertical dispersivity. This was a reasonable approach to take in view of the modeling objective, which was to compare three candidate sites and provide a travel-time rationale for choosing one over the others.

6.0 GROUNDWATER IMPACT ASSESSMENT

The potential groundwater impact at the proposed SALDS is addressed in this chapter. The following three aspects are considered: (1) geologic suitability; (2) hydrologic impacts; and (3) contaminant impacts. The impacts are based entirely on the modeling work described above, including the original input values and assumptions used by the modeling teams.

6.1 GEOLOGIC SUITABILITY

Stratigraphic studies at the SALDS suggest that nothing was encountered during the drilling of the two boreholes that would indicate the SALDS would perform any differently than any other site in the northern part of 200 West Area. The absence of a significant caliche zone in the Plio-Pleistocene unit suggests that there is no major barrier to the downward percolation of effluent through the sediment sequence. The sequence

is comprised predominantly of sandy silts that will retard the downward movement of liquid effluent when compared with more permeable surficial deposits such as sand and gravel which are common elsewhere on the Hanford Site.

The relative abundance of lower permeability sediments in the vicinity of the chosen SALDS provide slower groundwater seepage velocities than at many other potential locations on the Hanford Site. Slow rates of groundwater movement provide longer residence time for the radioactive decay of tritium to occur.

6.2 HYDROLOGIC IMPACTS

6.2.1 Lateral Spreading

The fine-grained nature of the Plio-Pleistocene sediments, as elsewhere in the 200 West Area, will cause some lateral spreading of effluent. Numerical modeling of flow and transport in the vadose zone confirms this (Lu et al. 1992), but suggests that the Plio-Pleistocene unit would not cause spreading to such a degree that would affect crib operation. Lateral spreading of the effluent is predicted to occur in the vadose zone at a depth of approximately 10 m (30 ft). The lateral spreading is caused by a low-conductivity layer at this depth. Vadose zone modeling suggests that lateral spreading of effluent at the Plio-Pleistocene unit boundary will not reach the inferred location of the erosional channel (Lu et al. 1992).

6.2.2 Groundwater Mounding

Gradual growth of a water table mound beneath the tritium discharge area is predicted by the vadose zone flow and solute transport modeling of Lu et al. (1992). After one year, the water table mound (at 95% saturation) reaches a maximum height of approximately 5 m (16 ft) above the preexisting water table. A steady-state saturation distribution is established after approximately 14 years, with a mound height slightly less than 5 m (16 ft). After discharge ceases the water table mound gradually decreases until it disappears about 5 years following the cessation of disposal at the SALDS (Lu et al. 1992).

6.2.3 Changes in the Groundwater Flow Directions

When the tritiated water disposed at the SALDS reaches the water table, it will flow in the direction of groundwater flow at the site. This is currently to the northeast.

A reversal of the natural groundwater flow direction (from northeast to southwest) is predicted to occur locally on the southwestern side of the SALDS as a result of the groundwater mound. As the mound decays following the cessation of disposal activities, the flow from southwest to northeast will eventually be reestablished. This is predicted to occur over approximately 5 years following the end of disposal to the SALDS.

6.3 POTENTIAL CONTAMINANT IMPACTS

The following sections describe potential impacts of tritiated water disposal to the soil column at the SALDS.

6.3.1 Vadose Zone

Tritiated water disposed in the crib will seep into the underlying sediments of the vadose zone immediately beneath the disposal crib. The tritium plume is predicted to reach the water table after approximately 1 year. At this time, maximum tritium concentrations in the vadose zone beneath the facility are equal to the source concentration decayed one year. For the 16.2 $\mu\text{Ci/L}$ used by Lu et al. (1992), this corresponds to a concentration of about 14 $\mu\text{Ci/L}$. The 14 $\mu\text{Ci/L}$ contour extends to a depth of approximately 25 m, and spreads laterally by about 4 m (13 ft) beyond the boundaries of the disposal crib. By 19 years, the maximum concentration contour is predicted to have reached a depth of 75 m (250 ft) which is 5 m (16 ft) below the original water table, with a lateral spread of 20 m (65 ft) in all directions. This configuration of tritium concentrations is predicted to be relatively stable for as long as disposal continues at a constant source concentration (Lu et al. 1992).

The reduced estimate of tritium levels in disposed effluents (5.4 to 6.3 $\mu\text{Ci/L}$) indicates that maximum tritium levels in the vadose zone will be on the order of 5 $\mu\text{Ci/L}$.

6.3.2 Suprabasalt Aquifer

Tritiated water from the SALDS is expected to reach the water table in the suprabasalt aquifer within 1 year of the start of disposal operations. Tritium concentrations are predicted to reach a maximum of 14 $\mu\text{Ci/L}$ (Lu et al. 1992). This is approximately 700 times the regulatory water quality criterion of 0.02 $\mu\text{Ci/L}$ (20,000 pCi/L) for tritium in groundwater in the State of Washington (WAC 173-200-040). However, lower estimate of tritium levels in the disposal effluent will reduce this figure to around 5 $\mu\text{Ci/L}$. The plume of tritiated water will move away from the SALDS under the prevailing groundwater flow system within the suprabasalt aquifer. Modeling has shown that travel time for the tritium plume to reach discharge points at the Columbia River is at least 100 years. The residence period for tritiated groundwater within the suprabasalt aquifer reduces tritium levels substantially (to less than 1/300 of the disposal level) as a result of radioactive decay.

6.3.3 Columbia Basalt Aquifers

The migration of tritiated groundwater from the SALDS into the deep Columbia Basalt aquifers underlying the Ringold Formation is not expected to occur. This expectation is primarily based on observations that upwelling from the deep basalt aquifers occurs in a general manner at the Hanford Site (DOE 1988). This upwelling occurs because the Pasco Basin forms the lowest point of the extensive Columbia Basalt aquifer system, and is primarily a discharge area through which regional groundwater reaches the Columbia River. Upward gradients can be locally reversed where groundwater mounds develop as a result of surface discharge. However, the downward gradients decay with increasing depth beneath the mounds. Contaminated groundwater

would have to be spread over the full thickness of the saturated suprabasalt aquifer for the plume to enter the basalt aquifer. Most information suggests that this is unlikely to occur.

A second mechanism which appears to reduce the amount of vertical dispersion occurring in plumes at the Hanford Site is anisotropy and the presence of sharp permeability contrasts near the water table. Where the water table occurs within the more permeable Hanford formation, this appears to provide a preferential groundwater flow pathway. This is evidenced by the relatively high concentrations of contaminants observed in springs discharging to the Columbia River.

6.3.4 Columbia River

Water disposed to the SALDS will eventually discharge into the Columbia River through groundwater seeps in the riverbed and springs along the riverbank. The concentration of tritium in the discharges has been simulated by groundwater modeling to reach a maximum of around 0.014 $\mu\text{Ci/L}$ (WHC 1992b). However, the predicted maximum tritium levels in groundwater discharging to the Columbia River may only be up to 0.0042 $\mu\text{Ci/L}$, as discussed in Section 5.2.8. Mixing and dilution are expected to reduce tritium levels within the receiving waters of the Columbia River to a fraction of the drinking water criterion (0.02 $\mu\text{Ci/L}$) within short distances downstream of the discharge points.

6.3.5 Pacific Ocean

The ultimate destination of the water discharged to the SALDS will be the Pacific Ocean at the mouth of the Columbia River. By this stage, mixing and dilution of the discharge water will mean that the SALDS will have an unmeasurable impact on seawater quality.

6.4 SEASONAL VARIATIONS

Natural recharge at the Hanford Site is small or zero (Gee et al. 1992). Seasonal changes in water levels caused by recharge are small to negligible. The proposed effluent disposal will operate continuously, with no significant variation related to the passing of the seasons. Seasonal variations in groundwater conditions will therefore be negligible.

6.5 CUMULATIVE EFFECTS

Short-term cumulative effects are expected to be negligible in view of the low levels of tritium currently observed at the proposed site. The most significant potential cumulative effects of the proposed disposal activity are institutional. They relate to the continued operation of B-Pond, the potential initiation of other waste water disposal facilities on the Hanford Site, and changes in influx from Cold and Dry Creek basins

resulting from changes in agricultural practices. Significant changes in any of these boundary conditions could affect the flow directions and travel times from the SALDS.

Other cumulative effects may potentially result from coalescence of the SALDS plume with the two existing tritium plumes identified in the 200 West area (see Figure 22):

- The northern plume appears to originate in the T Plant Area (DOE-RL 1992), and has relatively low tritium concentrations but appears to be moving northeast toward Gable Gap, and could potentially coalesce with the SALDS plume. However, concentrations of tritium in the SALDS plume are expected to be significantly higher than in the northern plume, such that any cumulative effect from coalescence is expected to be minimal.
- The southern plume appears to originate from the S Plant Area (DOE-RL 1992), and has higher tritium concentrations but appears to be moving mainly eastward. Mounding in the 200 East Area is expected to cause this plume to divide near the 200 East Area into northerly and southeasterly components which will migrate toward Gable Gap and the Columbia River, respectively. The largest component is expected to move southeast (DOE-RL 1992). Coalescence of the smaller northern component with the SALDS plume may eventually occur as both move through Gable Gap.

Plume migration is highly dependent on groundwater flow patterns, which are in turn dictated by natural and artificial recharge. These uncertainties, combined with the uncertainties of hydrogeologic modeling described in Section 5.2.8, limit the potential accuracy of predicted cumulative effects.

6.6 MONITORING CONSIDERATIONS

The modelling by Lu, et al. (1992) included the simulation of tritium levels at a monitoring well located 100 m (330 ft) downgradient of the facility (see Figure 18). The tritium plume is predicted to reach such a well after about 9 years. Tritium concentrations at the well are predicted to reach a steady-state maximum after 14 years. Lu et al. (1992) indicate that tritium concentrations at the monitoring well would decline rapidly once disposal operations are terminated.

7.0 CONCLUSIONS

The geohydrologic impacts of effluent disposed of at the SALDS will involve changes to groundwater levels, groundwater flow directions, and groundwater quality in

the suprabasalt aquifer system beneath the Hanford Site. Changes in groundwater level and flow direction will be local to the 200 West Area. Changes in groundwater quality will be minimized because of the extensive treatment given to the waste water prior to discharge. This report considers only changes in groundwater quality caused by high tritium levels in the disposal effluent.

Changes in flow directions within the aquifer will not be sufficient to affect the general direction of groundwater movement and plume advection away from the SALDS. Modeling indicates that the plume of tritiated water will move northeast from the site, passing through Gable Gap on its way to the Columbia River.

Tritiated water disposed at the SALDS is expected to have tritium levels of between 5.4 and 6.3 $\mu\text{Ci/L}$. A small reduction in tritium levels (about 10%) is expected by the time the effluent percolates through the vadose zone beneath the SALDS and reaches the water table at a depth of 70 m (230 ft) within the underlying aquifer. However, tritium levels in disposal water reaching the aquifer may still be on the order of 5 $\mu\text{Ci/L}$.

Radioactive decay of tritium will significantly reduce tritium concentrations by the time the plume reaches groundwater discharge points along the Columbia River. Travel times for first arrival at the Columbia River are at least 100 years, allowing at least eight half-life decay cycles. This will produce at least a 250-fold reduction in tritium levels. Dispersion of the tritiated water within the aquifer produces substantial additional reduction in tritium concentrations before natural discharge to the Columbia River.

Predictions of tritium concentrations in groundwater discharging to the Columbia River are based on generalized modeling work conducted for other purposes and which is strongly influenced by a series of limitations and assumptions. Some of these assumptions are inherently conservative, and lead to overestimates of the tritium concentrations. Also, the level of tritium assumed in the disposal effluent when the modeling was performed is significantly higher than current projections.

This geohydrological addendum to the 240 Engineering Report for Project C-018H has been assembled by a hydrogeologist from Golder Associates Inc., under the supervision of a Professional Engineer registered in the State of Washington. The material presented here includes original work not performed by Golder Associates Inc.

Engineer's Stamp

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Expires 2-21-94

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923-E026/45716/8-17-93

Figure 1. Hanford Site Location Map (adapted from PNL, 1988).

Legend

● Borehole Location
W7-3

923 E026.020/44878/8-17-93

Figure 2. Location of the State-Approved Land Disposal Site
(Reidel and Thornton 1992; Figure 1).

923-E026.020/23907/8-17-93

Figure 3. Structural Geologic Setting for the Hanford Site (Reidel and Thornton 1992).

923 E026.020/45720/8-17-93

Figure 4. Generalized Stratigraphy of the Suprabasalt Sediments at the Hanford Site (Lindsey 1991; Figure 5).

923 E026.020/44874/8-17-93

Figure 5. Generalized Composite Stratigraphy for the Ringold Formation from the Cold Creek Syncline and the Wahluke Syncline (Lindsey 1991).

923 E026.020/44875/8-17-93

Figure 6. Stratigraphic Units and Lithology Penetrated at the State-Approved Land Disposal Site by Borehole 699-48-77A (Reidel and Thornton 1992; Figure 6).

923 E026.020/44876/8-17-93

Figure 7. North-South Cross Section Through the Sedimentary Units Above the Columbia River Basalt Group at the State-Approved Land Disposal Site (Reidel and Thornton 1992; Figure 7).

923 E026.020/45721/8-17-93

Figure 8. Hanford Site Water Table Map, June 1990
(Newcomer et al. 1991).

923 E026.020/44877/8-17-93

Figure 9. Water Table Map for the State-Approved Land Disposal Site and Vicinity (Reidel and Thornton 1992 ; Figure 8).

Figure 10. Schematic Cross-Section of Vadose Zone Model
(adapted from Lu et al. 1992).

923 E026.020/45311/6-4-93

Figure 11. Hanford Site Groundwater Model Region (WHC 1992b; Figure 1).

923 E026.020/45312/6-4-93

Figure 12. Simulated 1944 Hydraulic Head Contours
(WHC 1992b; Figure 5).

923 E026.020/45313/6-4-93

Figure 13. Simulated 1979 Hydraulic Head Contours
(WHC 1992b; Figure 6).

923 E026.020/45325/8-17-93

Figure 14. State-Approved Land Disposal Site Source Representation for Plume Migration Modeling (WHC 1992b; Figure 7).

923 E026.020/45314/6-4-93

Figure 15. Simulated Hydraulic Head Contours with Disposal at the State-Approved Land Disposal Site (WHC 1992b; Figure 12).

923 E026.020/45315/6-4-93

Figure 16. Simulated Tritium Concentration Contours After 5 Years (WHC 1992b; Figure 21).

923 E026.020/45316/6-4-93

Figure 17. Simulated Tritium Concentration Contours After 55 Years (WHC 1992b; Figure 22).

923 E026.020/45317/6-4-93

Figure 18. Simulated Tritium Concentration Contours After 85 Years (WHC 1992b; Figure 23).

923 E026.020/45318/6-4-93

Figure 19. Simulated Tritium Concentration Contours After 105 Years (WHC 1992b; Figure 24).

923 E026.020/45319/6-4-93

Figure 20. Simulated Tritium Concentration Contours After 205 Years (WHC 1992b; Figure 25).

Source: WHC (1992b); Figure 26.

923 E026.020/45320/6-4-93

Figure 21. River Distances Used in Concentration Profiles, Figures 22 through 25 (WHC 1992b; Figure 26).

Figure 22. Tritium Concentrations in the Hanford Site Unconfined Aquifer in 1988 (from DOE-RL 1990).

See Map of River Distances on Figure 21.

Source: WHC (1992b); Figure 27.

923 E026.020/45321/6-4-93

Figure 23. Simulated Tritium Concentration Profile in Groundwater Discharging to the Columbia River After 105 Years (WHC 1992b; Figure 27).

923 E026.020/45322/6-4-93

Figure 24. Simulated Tritium Concentration Profile in Groundwater Discharging to the Columbia River After 125 Years (WHC 1992b; Figure 28).

Figure 25. Simulated Tritium Concentration Profile in Groundwater Discharging to the Columbia River After 165 Years (WHC 1992b; Figure 29).

923 E026.020/45323/6-4-93

923 E026.020/45324/6-4-93

Figure 26. Simulated Tritium Concentration Profile in Groundwater Discharging to the Columbia River After 205 Years (WHC 1992b; Figure 30).

Figure 27. Predicted Concentration of Tritium at a Downgradient Well as a Function of Time (Lu et al. 1992; Figure 18).

Table 1. Sediment Samples Collected for Analysis from Boreholes 699-48-77A and 699-48-77B.

HEIS sampler number	Depth	Comment
B01RGO (77B)	0.5 to 0.8 m (1.5-2.5 ft)	Vadose zone
B01RG1 (77B)	8.1 to 8.4 m (26.5-27.5 ft)	Vadose zone
B01RG2 (77B)	9.1 to 9.4 m (29.7-30.7 ft)	Vadose zone
B01RG3 (77A)	20.9 to 21.3 m (68.5-70.0 ft)	Vadose zone
B01RG4 (77A)	20.9 to 21.3 m (68.5-70.0 ft)	Vadose zone
B01RG5	--	QA/QC blank
B01RG6	--	QA/QC blank
B01RG7 (77A)	25.8 to 26.3 m (84.7-86.2 ft)	Vadose zone
B01RG8 (77A)	69.9 to 70.6 m (229.5-231.5 ft)	At water table
^a HEIS = Hanford Environmental Information System QA = quality assurance QC = quality control (77A) = Borehole 699-48-77A. (77B) = Borehole 699-48-77B.		

Table 2. Results of Sediment Analysis for Samples from Boreholes 699-48-77A and 699-48-77B. (Sheet 1 of 2)

Analyte	B01RG0	B01RG1	B01RG2	B01RG3	B01RG4	B01RG5*	B01RG6*	B01RG7	B01RG8
Aluminum	6,000	6,300	5,200	2,400	3,200	140	130	4,100	3,600
Antimony	6.4	5.6	4.7	<48	<45	<4.8	<4.8	6.6	<46
Arsenic	<4.6	<4.3	<4.5	<48	<45	<4.8	<4.8	<4.6	<46
Barium	70	56	42	40	36	2.2	2.2	60	30
Beryllium	0.29	0.32	0.24	<0.29	<0.27	<0.029	<0.29	0.20	<0.28
Cadmium	<0.28	<0.26	<0.27	<2.9	<2.7	<0.29	<0.029	0.55	<2.8
Calcium	5,700	5,200	20,000	3,800	4,200	36	39	4,500	2,300
Chromium	7.2	7.8	10	<9.7	<9.0	<0.97	<0.96	6.4	<9.3
Cobalt	8.2	7.0	7.0	<4.8	<4.5	0.51	<0.48	6.4	<4.6
Copper	12	12	15	7.4	11	<0.39	<0.38	11	6.8
Iron	17,000	14,000	14,000	7,900	9,800	200	210	12,000	8,400
Lead	5.1	4.7	4.8	<48	<45	<4.8	<4.8	5.9	<46
Lithium	6.3	5.3	6.6	<3.9	<3.6	<0.39	<0.38	4.5	4.3
Magnesium	3,300	2,800	3,600	1,900	2,500	29	18	2,900	2,500
Manganese	250	210	210	120	150	4.3	2.9	180	140
Molybdenum	<0.92	0.92	<0.91	<9.7	<9.0	<0.97	<0.96	<0.91	<9.3
Nickel	7.9	7.9	11	<9.7	11	<0.97	<0.96	10	<9.3
Phosphorus	730	630	820	640	740	<19	<19	1,200	400
Potassium	1,100	1,000	770	<580	<540	<58	66	700	<560
Selenium	<4.6	<4.3	<4.5	<48	<45	<4.8	<4.8	<4.6	<46
Silver	<0.55	<0.52	<0.54	<5.8	<5.4	<0.58	<0.58	<0.55	<5.6
Sodium	220	240	200	<19	38	23	21	220	170
Strontium	22	24	49	18	15	0.52	0.66	21	10

Table 2. Results of Sediment Analysis for Samples from Boreholes 699-48-77A and 699-48-77B. (Sheet 2 of 2)

Analyte	B01RG0	B01RG1	B01RG2	B01RG3	B01RG4	B01RG5*	B01RG6*	B01RG7	B01RG8
Uranium	6.5	<2.6	<2.7	<29	<27	11	<2.9	6.6	<28
Vanadium	40	38	31	13	18	<0.48	<0.48	31	15
Zinc	31	26	28	20	21	1.0	1.0	24	26
Bromide	<20	<20	<20	<20	<20	<20	<20	<20	<20
Phosphate	<20	<20	<20	<20	<20	<20	<20	<20	<20
pH	9.0	9.1	9.4	9.2	9.1	7.8	7.6	9.3	9.1
Fluoride	<2	<2	<2	<2.0	<2.0	<2.0	<2.0	<2.0	5
Cyanide	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
Chloride	<20	<20	<20	<20	<20	<20	<20	<20	<20
Nitrate	<20	<20	<20	<20	<0.2	<20	<0.2	<0.2	<20
Sulfate	<20	<20	37	<20	<20	<20	<20	<20	<20
Mercury	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Alpha	10.4	27.4	24.2	12.6	8.3	4.7	2.6	5.2	3.5
Beta	12.5	19.1	21.3	<10	<10	<10	<10	<10	<10
Cs-137	<2	<1E-5	<.01	<2	<2	<2	<1	<2	<2
* = Field blanks. All measurements in $\mu\text{g/g}$ except pH and radioactivity (pCi/g). Source: Reidel and Thornton (1992)									

Table 3. Sediment Leachate Test Information.

Test ID	Sediment#/Depth/Borehole	Leachate ID	HEIS no.
Run #1			
2-1190	CL#6/34.8-35.3'/77B	2-1190-01 2-1190-02 2-1190-03 2-1190-04	B01SJ5 B01SJ9 B01SK3 B01SK7
2-1192	CL#8/43.1-43.6'/77B	2-1192-01 2-1192-02 2-1192-03 2-1192-04	B01SJ6 B01SK0 B01SK4 B01SK8
2-1194	CL#10/57.2-57.7'/77B	2-1194-01 2-1194-02 2-1194-03 2-1194-04	B01SJ7 B01SK1 B01SK5 B01SK9
2-1195	CL#11/57.7-58.2'/77B	2-1195-01 2-1195-02 2-1195-03 2-1195-04	B01SJ8 B01SK2 B01SK6 B01SL0
Equipment blank		Eq. Blank	B01SL8
Deionized water blank--Run #1		DIH ₂ O-1	B01SL9
Run #2			
2-1185-01	CL#1/0.5'-1.0'/77B	2-1185-01 2-1185-02 2-1185-03 2-1185-04	B06V18 B06V22 B06V26 B06V30
2-1186	CL#2/1.5-2.0'/77B	2-1186-01 2-1186-02 2-1186-03 2-1186-04	B06V19 B06V23 B06V27 B06V31
2-1188	CL#4/26.0-26.5'/77B	2-1188-01 2-1188-02 2-1188-03 2-1188-04	B06V20 B06V24 B06V28 B06V32
2-1433	CL#12/68.0-68.5'/77B	2-1433-01 2-1433-02 2-1433-03 2-1433-04	B06V21 B06V25 B06V29 B01SL1
Deionized water blank - Run #2		DIH ₂ O-2	B01SL2
Source: Reidel and Thornton (1992) 77A = Borehole 699-48-77A. 77B = Borehole 699-48-77B.			

Table 4. Analytical Results for Sediments Leachate Samples.

Leachate ID	Element										
	pH	Ca	Mg	Na	K	Cl	F	NO ₃	PO ₄	SO ₄	SiO ₂
2-1190-01	8.0	41.0	7.0	8.8	6.4	28.3	0.5	6.7	<0.4	17	41.3
02	7.5	20.8	3.5	2.3	16.3	2.9	0.6	5.7	2.1	12	39.6
03	7.5	11.5	2.4	1.5	13.7	0.7	0.6	0.7	2.4	4	42.9
04	7.4	9.3	2.0	1.3	13.1	0.8	0.7	0.5	2.6	3	42.0
2-1192-01	8.0	39.6	10.8	11.2	6.8	2.3	0.5	1.7	<0.4	8	45.3
02	8.2	25.1	7.0	6.7	6.0	1.8	0.5	1.8	<0.4	15	44.8
03	8.0	19.2	5.3	5.5	5.3	0.7	0.6	0.8	<0.4	6	45.7
04	8.1	16.2	4.5	4.6	4.7	0.6	0.6	0.7	<0.4	4	44.6
2-1194-01	8.0	37.0	9.5	14.7	6.5	4.9	0.5	11.2	<0.4	41	46.7
02	8.2	22.6	5.9	5.1	5.2	0.8	0.5	1.2	<0.4	6	48.1
03	8.2	19.4	5.0	4.6	5.0	0.7	0.6	1.1	<0.4	3	50.3
04	8.1	15.6	4.0	3.8	4.3	0.7	0.6	1.0	<0.4	3	49.9
2-1195-01	8.0	33.8	8.7	7.3	6.5	4.8	0.6	4.2	<0.4	22	49.3
02	8.0	22.7	6.0	5.9	5.6	0.8	0.5	1.1	<0.4	4	48.8
03	8.0	20.7	5.3	5.1	5.3	0.7	0.6	4.1	<0.4	3	50.6
04	8.0	18.1	4.6	4.1	4.8	0.7	0.6	4.5	<0.4	2	50.4
Equipment Blank	5.6	1.2	0.2	1.6	0.2	0.8	<0.1	2.2	<0.4	2	0.1
Deionized H ₂ O-1	6.1	0.1	<0.1	<0.1	<0.1	<0.2	<0.1	<0.2	<0.4	<1	<0.1
2-1185-01	7.2	78.2	13.0	5.8	30.3	29.3	<0.3	42.7	<1.2	47	43.6
02	7.5	20.8	3.5	2.3	16.3	2.9	0.6	5.7	2.1	12	39.6
03	7.5	11.5	2.4	1.5	13.7	0.7	0.6	0.7	2.4	4	42.9
04	7.4	9.3	2.0	1.3	13.1	0.8	0.7	0.5	2.6	3	42.0
2-1186-01	8.0	37.2	4.8	5.8	1.7	2.4	0.5	6.8	<0.4	13	56.9
02	7.9	24.7	3.1	4.3	1.2	0.7	0.6	1.7	<0.4	4	55.8
03	7.8	22.1	2.8	3.7	1.2	0.7	0.6	1.0	<0.4	3	55.0
04	7.8	20.2	2.4	2.9	1.2	0.7	0.6	0.7	<0.4	3	53.2
2-1188-01	7.3	47.2	7.9	15.8	7.4	55.4	0.4	14.1	<0.8	18	34.1
02	7.6	18.2	3.0	9.2	4.7	1.2	0.5	2.0	<0.4	4	38.8
03	7.8	16.3	2.5	8.3	4.5	0.6	0.7	0.7	<0.4	4	38.8
04	7.8	12.6	2.0	6.7	4.3	0.6	0.7	0.6	<0.4	2	45.4
2-1433-01	7.8	28.0	6.0	8.4	5.3	4.3	0.7	14.1	<0.8	14	43.3
02	7.9	15.2	3.4	5.8	4.1	0.7	0.5	2.6	<0.4	3	42.2
03	8.2	12.6	2.9	4.9	3.8	0.6	0.4	1.4	<0.4	3	42.1
04	8.3	10.5	2.4	3.9	3.3	0.6	0.3	0.9	<0.4	2	39.8
Deionized H ₂ O-2	6.3	0.1	<0.1	<0.1	<0.1	<0.2	<0.1	<0.2	<0.4	<1	<0.1

Source: Reidel and Thornton (1992)

Notes: Results (except pH) are expressed in mg/L

Table 5. Contamination Indicator Parameters Used for Borehole 699-48-77A. (Page 1 of 4)

Constituent Name			Lab	DWS		Number of Samples		
Short (Method)	Full	Units	CRQL	Limit	Agency	Total	>CRQL	>DWS
CONTAMINATION INDICATOR PARAMETERS								
CONDFLD	Conductivity, field	μS	-	700	WDOE	1	1	0
TOC	Total Organic Carbon	$\mu\text{g/L}$	1000	-	-	1	0	-
TOXLDL	Total Organic Halogen	$\mu\text{g/L}$	10	-	-	1	1	-
PH-FLD	pH, field	-	6.5-8.5	-	EPAS	1	1	0
DRINKING WATER PARAMETERS								
a-BHC	alpha-BHC	$\mu\text{g/L}$	0.05	4	EPA	1	0	0
ARSENIC	Arsenic	$\mu\text{g/L}$	5	50	EPA	1	0	0
PARSENI	Arsenic, filtered	$\mu\text{g/L}$	5	50	EPA	1	0	0
BARIUM	Barium	$\mu\text{g/L}$	20	1,000	EPA	1	1	0
FBARIUM	Barium, filtered	$\mu\text{g/L}$	20	1,000	EPA	1	1	0
b-BHC	Beta-BHC	$\mu\text{g/L}$	0.05	4	EPA	1	0	0
CADMIUM	Cadmium	$\mu\text{g/L}$	10	10	EPA	1	0	0
FCADMIU	Cadmium, filtered	$\mu\text{g/L}$	10	10	EPA	1	0	0
CHROMUM	Chromium	$\mu\text{g/L}$	20	50	EPA	1	0	0
FCHROMI	Chromium, filtered	$\mu\text{g/L}$	20	50	EPA	1	0	0
d-BHC	Delta-BHC	$\mu\text{g/L}$	0.1	4	EPA	1	0	0
ENDRIN	Endrin	$\mu\text{g/L}$	0.1	0.2	EPA	1	0	0
FLOURID	Fluoride	$\mu\text{g/L}$	100	4,000	EPA	1	1	0
ALPHA	Gross alpha	pCi/L	4	15	EPA	2	1	0
BETA	Gross beta	pCi/L	8	50	EPA	2	1	0
LEAD	Lead	$\mu\text{g/L}$	5	50	EPA	1	0	0
FLEAD	Lead, filtered	$\mu\text{g/L}$	5	50	EPA	1	0	0
MERCURY	Mercury	$\mu\text{g/L}$	0.2	2	EPA	1	0	0
FMERCUR	Mercury, filtered	$\mu\text{g/L}$	0.2	2	EPA	1	0	0
METHLOR	Methoxychlor	$\mu\text{g/L}$	2	100	EPA	1	0	0
SELENIUM	Selenium	$\mu\text{g/L}$	10	10	EPA	1	0	0
FSELENI	Selenium, filtered	$\mu\text{g/L}$	10	10	EPA	1	0	0
SILVER	Silver	$\mu\text{g/L}$	20	50	EPA	1	0	0
FSILVER	Silver, filtered	$\mu\text{g/L}$	20	50	EPA	1	0	0
TOXAENE	Toxaphene	$\mu\text{g/L}$	2	5	EPA	1	0	0
LINDANE	gamma-BHC (Lindane)	$\mu\text{g/L}$	0.05	4	EPA	1	0	0

Table 5. Contamination Indicator Parameters Used for Borehole 699-48-77A. (Sheet 2 of 4)

Constituent Name			Lab	DWS		Number of Samples		
Short (Method)	Full	Units	CRQL	Limit	Agency	Total	>CRQL	>DWS
GROUNDWATER QUALITY PARAMETERS								
CHLORID	Chloride	µg/L	200	250,000	EPAS	1	1	0
IRON	Iron	µg/L	20	300	EPAS	1	1	1
FIRON	Iron, filtered	µg/L	20	300	EPAS	1	0	0
MANGESE	Manganese	µg/L	10	50	EPAS	1	1	0
FMANGAN	Manganese, filtered	µg/L	10	50	EPAS	1	0	0
PHENOL	Phenol	µg/L	10	-	-	1	0	-
SODIUM	Sodium	µg/L	300	-	-	1	1	-
FSODIUM	Sodium, filtered	µg/L	300	-	-	1	1	-
SULFATE	Sulfate	µg/L	500	250,000	EPAS	1	1	0
SITE SPECIFIC AND OTHER CONSTITUENTS								
1,1,1-T	1,1,1-Trichloroethane	µg/L	5	200	EPA	1	0	0
1,1,2-T	1,1,2-Trichloroethane	µg/L	5	-	-	1	0	-
1,1-DIC	1,1-Dichloroethane	µg/L	5	-	-	1	0	-
1,2-DIC	1,2-Dichloroethane	µg/L	5	5	EPA	1	0	0
14-dben	1,4-Dichlorobenzene	µg/L	5	750	EPA	1	0	0
BUTANOL	1-Butanol	mg/L	1	-	-	1	0	-
2METHPH	2-Methylphenol	µg/L	10	-	-	1	0	-
DDD	4,4'-DDD	µg/L	0.1	-	-	1	0	-
DDE	4,4'-DDE	µg/L	0.05	-	-	1	0	-
DDT	4,4'-DDT	µg/L	0.1	-	-	1	0	-
MIBK	4-Methyl-2-pentanone	µg/L	50	-	-	1	0	-
4METHPH	4-Methylphenol	µg/L	10	-	-	1	0	-
ACETONE	Acetone	µg/L	100	-	-	1	0	-
ALDRIN	Aldrin	µg/L	0.05	-	-	1	0	-
ANTIONY	Antimony	µg/L	200	-	-	1	0	-
FANTIMO	Antimony, filtered	µg/L	200	-	-	1	0	-
BENZENE	Benzene	µg/L	5	5	EPA	1	0	0
BERYLUM	Beryllium	µg/L	3	-	-	1	0	-
FBERYLL	Beryllium, filtered	µg/L	3	-	-	1	0	-
BROMIDE	Bromide	µg/L	500	-	-	1	0	-
CALCIUM	Calcium	µg/L	100	-	-	1	1	-
FCALCTU	Calcium, filtered	µg/L	100	-	-	1	1	-

Table 5. Contamination Indicator Parameters Used for Borehole 699-48-77A. (Sheet 3 of 4)

Constituent Name			Lab	DWS		Number of Samples		
Short (Method)	Full	Units	CRQL	Limit	Agency	Total	>CRQL	>DWS
TETRANE	Carbon tetrachloride	µg/L	5	5	EPA	1	0	0
CHLOANE	Chlordane	µg/L	0.1	2	EPAP	1	0	0
CHLFORM	Chloroform	µg/L	5	100	EPA	1	0	0
COBALT	Cobalt	µg/L	20	-	-	1	0	-
FCOBALT	Cobalt, filtered	µg/L	20	-	-	1	0	-
COLI-MFT	Coliforms, membrane filter	No.	1	1	EPA	2	2	2
COPPER	Copper	µg/L	20	1,000	EPAS	1	0	0
FCOPPER	Copper, filtered	µg/L	20	1,000	EPAS	1	0	0
CYANIDE	Cyanide	µg/L	20	-	-	1	0	-
DECANE	Decane	µg/L	10	-	-	1	0	-
DIELRIN	Dieldrin	µg/L	0.05	-	-	1	0	-
DODECAN	Dodecane	µg/L	10	-	-	1	0	-
ENDO1	Endosulfan I	µg/L	0.1	-	-	1	0	-
ENDO2	Endosulfan II	µg/L	0.05	-	-	1	0	-
ENDSFAN	Endosulfane sulfate	µg/L	0.5	-	-	1	0	-
ENDRALD	Endrin Aldehyde	µg/L	0.2	-	-	1	0	-
HEPTLOR	Heptachlor	µg/L	0.05	0.4	EPAP	1	0	0
HEPTIDE	Heptachlor epoxide	µg/L	1	-	-	1	0	-
HYDRAZI	Hydrazine	µg/L	30	-	-	1	0	-
I-129LD	Iodine-129, low detection	pCi/L	1	1	EPAR	1	0	0
MAGNES	Magnesium	µg/L	100	-	-	1	1	-
FMAGNES	Magnesium, filtered	µg/L	100	-	-	1	1	-
METHONE	Methyl ethyl ketone	µg/L	100	-	-	1	0	-
METHYCH	Methylene chloride	µg/L	5	-	-	1	0	-
NAPHTHA	Naphthalene	µg/L	10	-	-	1	0	-
NICKEL	Nickel	µg/L	30	-	-	1	0	-
FNICKEL	Nickel, filtered	µg/L	30	-	-	1	0	-
NITRATE	Nitrate	µg/L	200	45,000	EPA	1	1	0
NITRITE	Nitrite	µg/L	200	3,300	EPAP	1	0	0
PENTCHP	Pentachlorophenol	µg/L	50	200	EPAP	1	0	0
PHOSPHA	Phosphate	µg/L	400	-	-	1	0	-
POTASUM	Potassium	µg/L	300	-	-	1	1	-
FPOTASS	Potassium, filtered	µg/L	300	-	-	1	1	-
TMP-C	Temperature, field	Deg C	-	-	-	1	1	-

Table 5. Contamination Indicator Parameters Used for Borehole 699-48-77A. (Sheet 4 of 4)

Constituent Name			Lab	DWS		Number of Samples		
Short (Method)	Full	Units	CRQL	Limit	Agency	Total	>CRQL	>DWS
PERCENE	Tetrachloroethene	µg/L	5	5	EPAP	1	0	0
TETRADE	Tetrachloroethene	µg/L	10	-	-	1	0	-
TAF	Tetrahydrofuran	µg/L	10	-	-	1	0	-
TIN	Tin	µg/L	100	-	-	1	0	-
FTIN	Tin, filtered	µg/L	100	-	-	1	0	-
TOLUENE	Toluene	µg/L	5	2,000	EPAP	1	0	0
TRIBUTIPH	Tributyl Phosphate	µg/L	10	-	-	1	0	-
TRICENE	Trichloroethene	µg/L	5	5	EPA	1	0	0
TRITIUM	Tritium	pCi/L	500	20,000	EPAR	2	0	0
VANADUM	Vanadium	µg/L	30	-	-	1	0	-
FVANADI	Vanadium, filtered	µg/L	30	-	-	1	0	-
VINYIDE	Vinyl chloride	µg/L	10	2	EPA	1	0	1*
XYLENE	Xylenes (total)	µg/L	5	-	-	1	0	-
ZINC	Zinc	µg/L	10	5,000	WDOE	1	1	0
FZINC	Zinc, filtered	µg/L	10	5,000	WDOE	1	0	0
MCRESOL	m-Cresol	µg/L	10	-	-	1	0	-
TRANDCE	trans-1,2-Dichloroethylene	µg/L	5	100	EPAP	1	0	0

Source: Reidel and Thornton (1992)

DWS = Drinking Water Standard

CRQL = contract-required quantification limits

EPAS = Environmental Protection Agency, Secondary Standard

EPAP = Environmental Protection Agency, Pesticide Standard

EPAR = Environmental Protection Agency, Radionuclide

WDOE = Washington State Department of Ecology

µg/L = micrograms per liter

BHC = benzene hexachloride

4,4'-DDD = 4,4-dichlorodiphenyldichloroethane

4,4'-DDE = 4,4-dichlorodiphenyldichloroethylene

4,4'-DDT = 4,4-dichlorodiphenyltrichloroethane

EPA = Environmental Protection Agency, Primary Standard

Table 6. Parameters with at least One Detected Value for Groundwater Analyses from Borehole 699-48-77A.

Analyte	Units	Sample Number/Data Collected			
		B06W97 06/19/92	B06W98F 06/19/92 Filtered	B078V0 09/01/92	B078V0 09/02/92
Barium	(mg/L)	0.074	0.067	-	-
Calcium	(mg/L)	33	31	-	-
Chloride	(mg/L)	6.6	-	-	-
Fluoride	(mg/L)	0.4	-	-	-
Iron	(mg/L)	0.56	0.02 U	-	-
Magnesium	(mg/L)	12	12	-	-
Manganese	(mg/L)	0.027	0.01 U	-	-
Nitrate	(mg/L)	28	-	-	-
Potassium	(mg/L)	3.7	3.3	-	-
Sodium	(mg/L)	17	15	-	-
Sulfate	(mg/L)	25	-	-	-
Zinc	(mg/L)	0.087	0.01 U	-	-
Temperature	(deg C)	18.5	-	-	-
Alpha	(pCi/L)	1.73 U	-	1.76	1.76
Beta	(pCi/L)	5.05 U	-	-	2.7
Coliform	(per dL)	65	-	2	2
Field Conductivity	(μ S)	316			
Field pH	(pH)	7.71			
TOC	(mg/L)	1.00 U			
TOX	(μ g/L)	20.0			
Source: Reidel and Thornton (1992) TOC = Total Organic Carbon TOX = Total Organic Halogens					

Table 7. Material Properties for Vadose Zone Modeling of the SALDS,
Project C-018H.

Material Type #	6	5	4	3	2	1
Formation type	Plio-Pleistocene	Plio-Pleistocene	Plio-Pleistocene	Plio-Pleistocene	Upper Ringold	Middle Ringold
Soil Type	Gravelly sand	Sand & gravel	Coarse sand	Sand	Sand	Sand & gravel
Sample #	2-1169	2-1170	2-1176	2-1181	2-1431	2-1432
Well #	699-48-77B	699-48-77B	699-48-77B	699-48-77B	699-48-77A	699-48-77A
Depth (ft)	26.5-27.0	29.2-29.7	42.6-43.1	46.3-46.8	67.5-68.0	90.5-91.0
Depth (m)	8.08-8.23	8.90-9.05	12.98-13.14	14.11-14.26	20.57-20.73	27.58-27.74
θ_r dry	0.057	0.049	0.033	0.053	0.054	0.037
θ_s sat	0.3732	0.3563	0.3548	0.2839	0.3603	0.2960
ρ_{bulk} (g/cm ³)	1.80	1.71	NA	1.76	1.80	1.63
SG (g/cm ³)	2.72	2.62	2.68	2.74	2.67	2.71
Porosity (%)	33.80	31.66	NA	35.77	32.91	39.72
α_{dry} (1/cm)	0.0101	0.0083	0.0407	0.0521	0.0399	0.0155
α_{dry} (1/m)	1.01	0.83	4.07	5.21	3.99	1.55
n	2.3808	1.6101	1.9286	1.7171	2.0770	1.5156
m = 1-1/n	0.5800	0.3790	0.4815	0.4176	0.5104	0.3402
K_s (cm/s)	5.30 E-3	1.29 E-4	1.96 E-2	8.20 E-3	1.80 E-2	1.37 E-2
K_s (m/yr)	1671.4	40.7	6181.1	2586.0	5676.5	4320.4

Source: Lu et al. (1992)

 θ = moisture content ρ = density K_s = saturated hydraulic conductivity α, n, m = curve-fitting parameters for moisture retention/hydraulic conductivity characteristics.

SG = specific gravity.