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Groundwater Impact Assessment Report for the 1325-N Liquid Waste Disposal Facility

Prepared for the U.S. Department of Energy
Office of Environmental Restoration and
Waste Management

Westinghouse
Hanford Company Richland, Washington

Hanford Operations and Engineering Contractor for the
U.S. Department of Energy under Contract DE-AC06-87RL10930

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EXECUTIVE SUMMARY

The 105-N and 109-N Facilities discharged effluent to the 1325-N Liquid Waste Disposal Facility (1325-N LWDF) located east of the 100-N Area perimeter fence. This report satisfies one of the requirements of the *Hanford Federal Facility Agreement and Consent Order* (Tri-Party Agreement) Milestone M-17-00B as agreed by the U.S. Department of Energy, the Washington State Department of Ecology, and the U.S. Environmental Protection Agency.[1] Tri-Party Agreement Milestone M-17-00B includes a requirement to assess impacts to groundwater from disposal of the 105-N Reactor and 109-N Heat Exchanger Buildings effluent to the 1325-N LWDF. In addition, the 1325-N LWDF is a *Resource Conservation and Recovery Act of 1976*[2] treatment, storage, and disposal facility covered by the *Closure/Post-Closure Plan for the 1301-N and 1325-N Liquid Waste Disposal Facilities*.[3]

There is groundwater contamination, primarily strontium-90, tritium, and sulfate, in the unconfined aquifer beneath the 100-N Area. The contaminant plumes are a result of past-practice reactor and disposal operations in the

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[1]Ecology, EPA, and DOE, 1991, *Hanford Federal Facility Agreement and Consent Order*, Second amendment, 2 vols, 89-10, Rev. 2, Washington State Department of Ecology, U.S. Environmental Protection Agency, and U.S. Department of Energy, Olympia, Washington.

[2]*Resource Conservation and Recovery Act of 1976*, 42 USC 6901, et seq.

[3]Diediker, L. A., and J. A. Hall, 1987, *Closure/Post-Closure Plan for the 1301-N and 1325-N Liquid Waste Disposal Facilities*, UNI-3533, UNC Nuclear Industries, Inc., Richland, Washington.

100-N Area currently being investigated as part of the 100-NR-1 and 100-NR-2 Operable Units.[4,5]

One-dimensional or unit-gradient flow and transport calculations suggest that strontium-90 reached the water table during the operational period of use for the 1325-N LWDF. Continuing migration of strontium-90 into the aquifer may be occurring due to natural recharge and water table fluctuations. The estimated magnitude of additional contaminant input to the aquifer caused by hypothetical slug releases of spent emergency water to the crib is about the same order of magnitude as the annual contaminant input to the aquifer caused by natural recharge. The localized concentrations caused by both types of input exceed the Washington State Water Quality Standard of 8 pCi/L and are potentially significant. Thus some mitigating measures for controlling the natural infiltration around the crib (e.g., restoration of natural vegetation around the crib or installation of an interim infiltration barrier) should be investigated as part of ongoing remediation efforts. Furthermore, emergency water should not be discharged to the crib.

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[4]DOE-RL, 1991a, *RCRA Facility Investigation/Corrective Measures Study Work Plan for the 100-NR-1 Operable Unit, Hanford Site, Richland, Washington*, DOE/RL-91-46, U.S. Department of Energy, Richland Operations Office, Richland, Washington.

[5]DOE-RL, 1991b, *RCRA Facility Investigation/Corrective Measures Study Work Plan for the 100-NR-2 Operable Unit, Hanford Site, Richland, Washington*, DOE/RL-90-22, U.S. Department of Energy, Richland Operations Office, Richland, Washington.

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LIST OF TERMS

BTV	background threshold value
CERCLA	*Comprehensive Environmental Response, Compensation, and Liability Act of 1980*
CRBG	Columbia River Basalt Group
CRQL	contractually required quantification limit
DOE	U.S. Department of Energy
Ecology	Washington State Department of Ecology
EPA	U.S. Environmental Protection Agency
FL(LP)	low-pressure flush line
LWDF	Liquid Waste Disposal Facility
ppb	parts per billion
ppm	parts per million
RCRA	*Resource Conservation and Recovery Act of 1976*
RDR	radioactive drain
RL	DOE, Richland Operations Office
1324-N	1324-N Surface Impoundment
Tri-Party Agreement	*Hanford Federal Facility Agreement and Consent Order*
WAC	*Washington Administrative Code*
WHC	Westinghouse Hanford Company
WWQS	Washington State Water Quality Standards

GROUNDWATER IMPACT ASSESSMENT REPORT FOR THE 1325-N LIQUID WASTE DISPOSAL FACILITY

1.0 INTRODUCTION

Groundwater impact assessments are required for a number of liquid effluent receiving sites in accordance with the *Hanford Federal Facility Agreement and Consent Order* (Tri-Party Agreement) Milestones M-17-00A and M-17-00B, as agreed by the U.S. Department of Energy (DOE), the Washington State Department of Ecology (Ecology), and the U.S. Environmental Protection Agency (EPA) (Ecology et al. 1991). This report assesses the impacts to groundwater from the disposal of past and present effluent from various 100-N Area facilities to the 1325-N Liquid Waste Disposal Facility (1325-N LWDF).

1.1 BACKGROUND

In response to public comments on the original Tri-Party Agreement and at the request of the signatories on the Tri-Party Agreement, the DOE, Richland Operations Office (RL) conducted a study to assess the impact of liquid effluents discharged to the ground at the Hanford Site (WHC 1990a, 1990b). The EPA and Ecology expressed several concerns regarding uncertainties in the evaluations made by RL. Foremost among these concerns were the lack of site-specific data, the need to consider interactions with adjacent liquid discharge facilities, and the need for more rigorous models of contaminant transport. As a result of these concerns, the RL, Ecology, and EPA (the three parties) created a series of Tri-Party Agreement milestones, including M-17-00A, M-17-00B, M-17-13, and M-17-13A, which pertain to groundwater impact assessments.

Tri-Party Agreement Milestones M-17-00A and M-17-00B require impact assessments for Phase I and II waste streams. Phase I and II waste streams are defined in Stordeur and Flyckt (1988). The 1325-N LWDF was defined as a Phase I waste stream. Tri-Party Agreement Milestone M-17-13 required the development of a methodology for assessing the impact of liquid effluent discharge on groundwater, which resulted in the document *A Methodology for Assessing Impacts to Groundwater from Disposal of Liquid Effluent to the Soil at the Hanford Site* (Tyler 1991). Thirty days after regulatory approval of the methodology document, as required by Tri-Party Agreement Milestone M-17-13A, a schedule for performing assessments at 13 receiving sites was completed. A draft *Resource Conservation and Recovery Act of 1976* (RCRA) closure/post-closure plan has been prepared for the 1325-N LWDF (Diediker and Hall 1987) and is scheduled to be resubmitted in June 1994 (Tri-Party Agreement Milestone M-20-31). Initial characterization of this site is presently in progress.

1.2 METHODOLOGY

The methodology document by Tyler (1991) was followed in preparing the groundwater impact assessment for the 1325-N LWDF. Included in that document is the categorization of each of the 13 receiving sites into 1 of 3 levels,

based on the amount of effort needed to perform the assessment. A level 1 receiving site groundwater impact assessment relies on available information. A groundwater impact assessment of a level 2 receiving site may require nonintrusive field work to verify the extent of existing contamination. A level 3 site may require intrusive field work. If it is discovered that existing information is inadequate through the course of performing a level 1 impact assessment, the assessment may be raised to a level 2 or 3.

The methodology document outlines several tasks to be conducted as part of the groundwater impact assessment for level 1 receiving sites:

- Prepare and present a plan describing how the groundwater impact assessment will be conducted
- Characterize the liquid effluent stream
- Evaluate the site-specific hydrogeology
- Develop a site conceptual model
- Assess the hydrologic impact of the liquid effluent stream
- Assess the contaminant impact of the liquid effluent stream
- Evaluate the adequacy of the existing monitoring well network
- Prepare a written report of the results.

The tasks required for level 2 and 3 receiving sites include those outlined above (with minor differences) and include field work-related activities as follows:

- Organize and plan facility-specific assessment (same as level 1 tasks)
- Characterize effluent using existing data (same as level 1 tasks)
- Develop a preliminary site conceptual model--aids in identification of information necessary to refine the conceptual model and support assessment techniques that may have more rigorous data requirements
- Identify additional information needs and plan data collection--may include collection of field data and laboratory analyses (water chemistry sampling, shallow soil samples, remote sensing, geophysical surveys, water level measurements, and physical and chemical analyses of archived soil samples)
- Level 3 data collection may also include borehole drilling and sampling, monitoring well construction (RCRA standard), aquifer testing, and more extensive data collection for activities listed in the previous bullet
- Collect field data
- Analyze and interpret data to refine conceptual model

- Assess groundwater impacts and identify additional information needs
- Prepare a written report of the results.

In the methodology report (Tyler 1991), the 1325-N LWDF was categorized as a level 1 receiving site because of the substantial decreases in allowable discharge in accordance with the Tri-Party Agreement flow restrictions (Tyler 1991). An administrative policy to cease discharges, except for emergency fire flow, was implemented in April 1991. New monitoring wells were installed at this site to characterize the groundwater in support of RCRA closure/post-closure activities (Hartman 1992). In addition, this site is part of the 100-NR-2 Groundwater Operable Unit and the 100-NR-1 Source Operable Unit, which are controlled by the RCRA Facility Investigation/ Corrective Measures Study Work Plans (DOE-RL 1991a, 1991b). This impact assessment incorporated data collected under the RCRA Facility Investigation/ Corrective Measures Study Work Plans.

Several key assumptions inherent to all groundwater impact assessments are explained in the methodology document (Tyler 1991) and warrant summarizing here. For this impact assessment, the following assumptions are relevant.

- The expected level of impact from the use of the receiving site determines how well the chemistry, geology, and hydrology need to be understood.
- Modeling sophistication is tailored to available information and the expected level of impact to the receiving site.
- New site-specific geologic, hydrologic, and groundwater data obtained through drilling require 1.5 to 2 years to acquire.
- Data collection and modeling activities are integrated with other Tri-Party Agreement milestones.
- Existing data are treated as fully validated.

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2.0 100-N AREA FACILITIES DESCRIPTION

2.1 LOCATION

The Hanford Site is a 1,450-km^2 (560-mi^2) tract of land located in Benton, Franklin, and Grant Counties in the south-central portion of Washington State. The 100-N Area is located along the south shore of the Columbia River in the northern part of the Hanford Site, approximately 74 km (46 mi) north of the city of Richland (Figure 1). The 1325-N LWDF is located approximately 18 m (60 ft) above and 732 m (2,400 ft) east of the Columbia River and outside the 100-N Area perimeter fence (Figure 2).

2.2 HISTORY

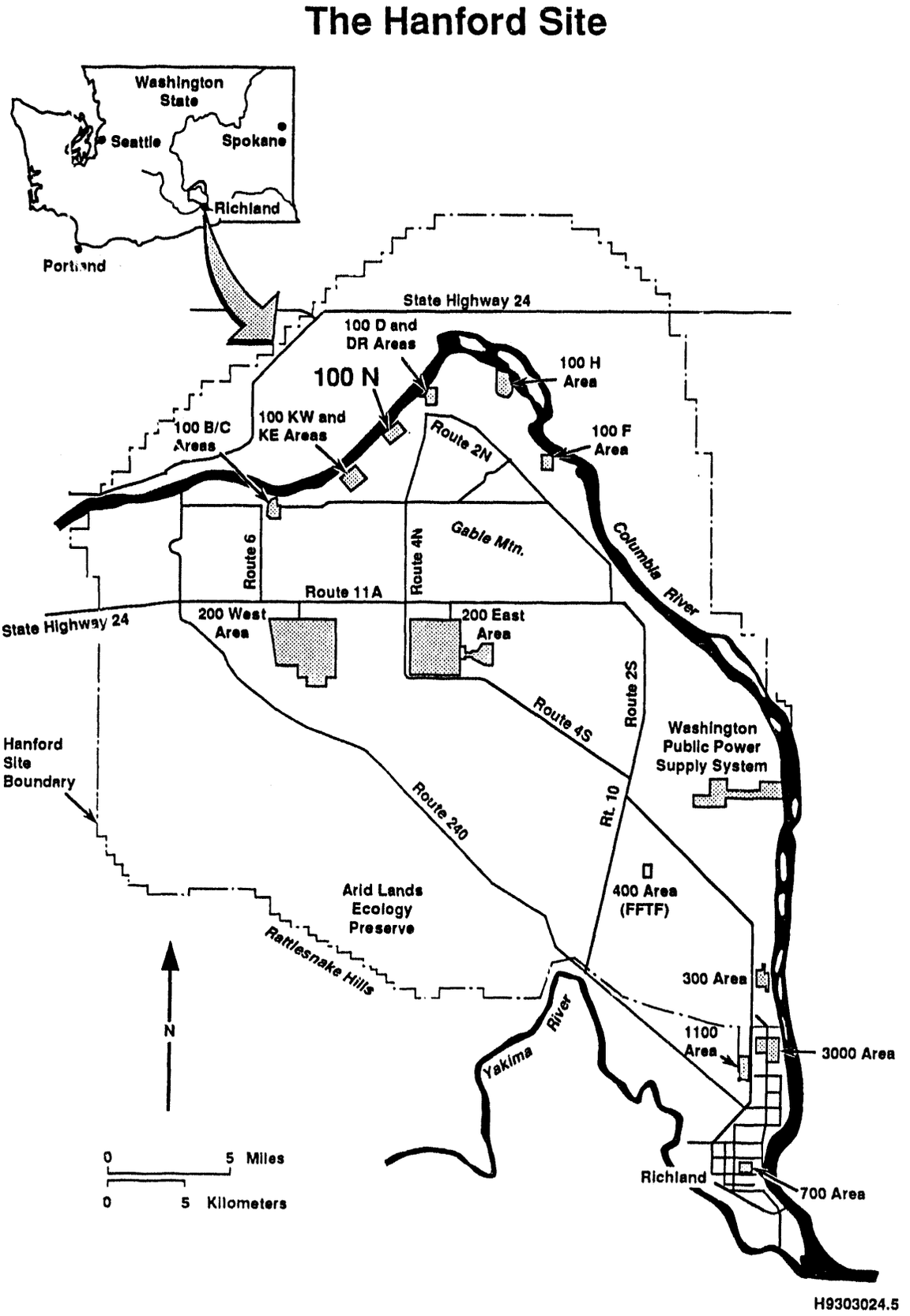
In 1943 the Hanford Site was chosen as a location for the Manhattan Project to produce plutonium for use in nuclear weapons. The 100-N Area at Hanford was used from 1963 to 1987 for a dual-purpose, plutonium production and steam generation reactor and related operational support facilities (Diediker and Hall 1987). In November 1989, the reactor was put into dry layup status. During operations, chemical and radioactive wastes were released into the area soil, air, and groundwater. The 1325-N LWDF was constructed in 1983 to replace the 1301-N Liquid Waste Disposal Facility (1301-N LWDF). The two facilities operated simultaneously from 1983 to 1985. The 1301-N LWDF was retired from use in 1985 and the 1325-N LWDF continued operation until April 1991, when active discharges to the facility ceased. Effluent discharge to the piping system has been controlled by administrative means.

2.3 FACILITIES

2.3.1 1325-N LWDF

The 1325-N LWDF is a crib and trench facility (Figure 3). The crib is 73 by 76 by 5 m (240 by 250 by 15 ft) and consists of a drain field pipe system covered with precast, prestressed concrete panels (Figure 4). The cover is 3 m (10 ft) below ground surface and 1.5 m (5 ft) above the percolation surface (see Figure 3, inset). The trench is 914 by 17 by 2 m (3,000 by 55 by 7 ft) and ties into the crib at its northern and eastern corners (effluent from each corner combines in a common weir box) (see Figures 3 and 4). The 1325-N LWDF began operation in 1983 as a crib that was constructed to replace the 1301-N LWDF. In 1985, the extension trench was added to the facility to increase its operational capacity (DOE-RL 1991a) and the 1301-N LWDF was shut down. The facility received effluent until April 1991, when all discharges were stopped through the use of administrative controls.

Figure 1. Location of the 100-N Area on the Hanford Site.



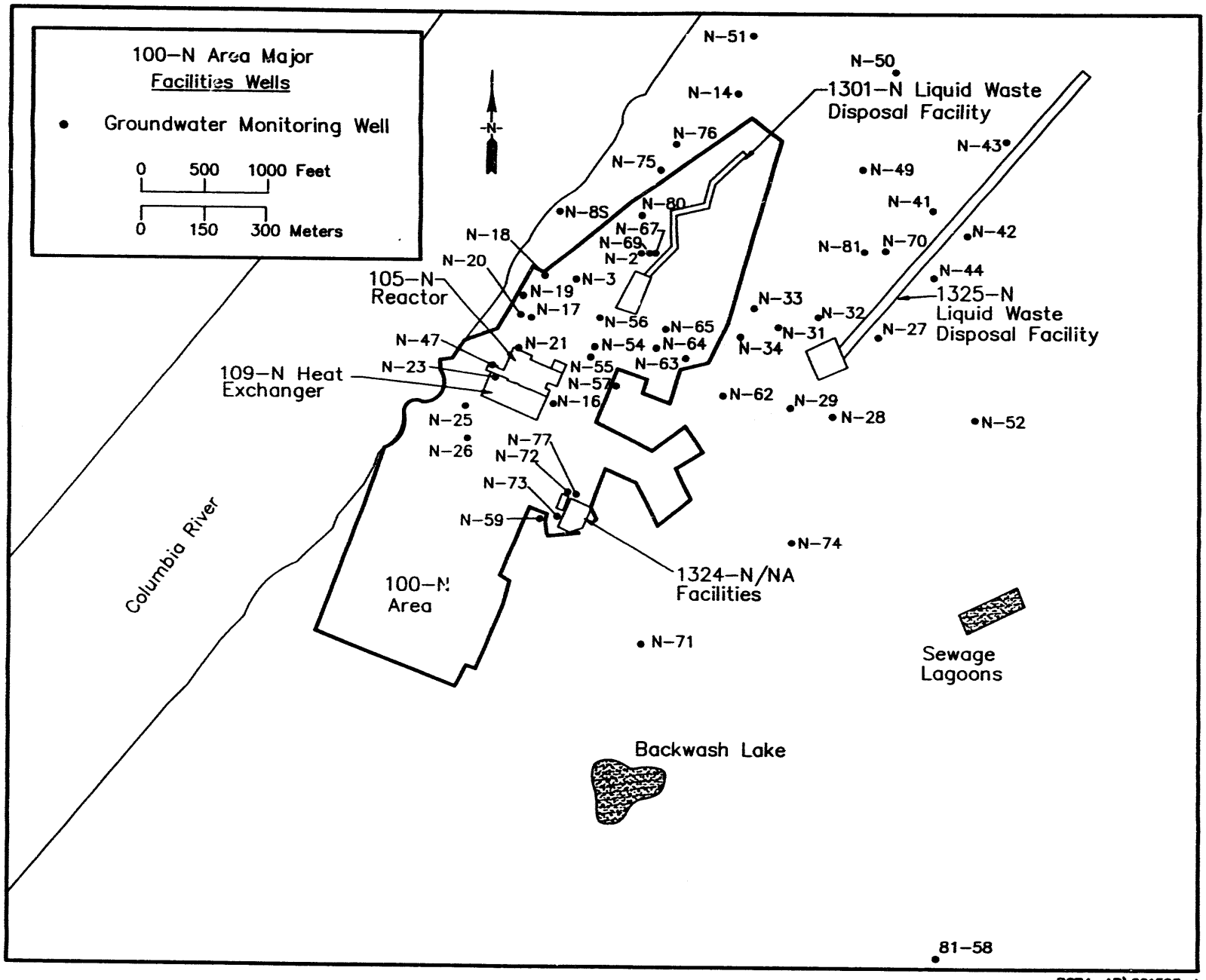


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Figure 2. Location of Major 100-N Area Facilities.

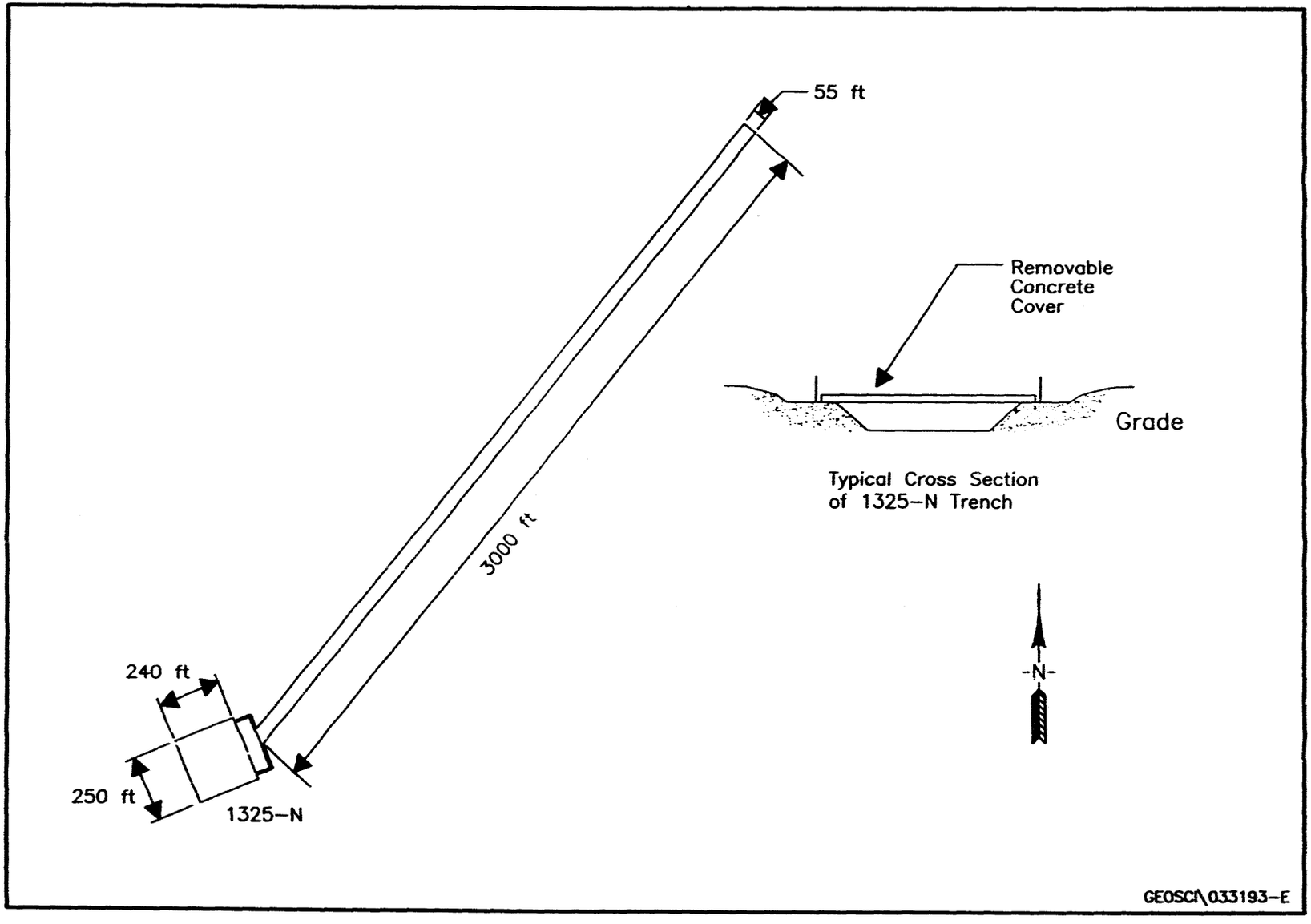
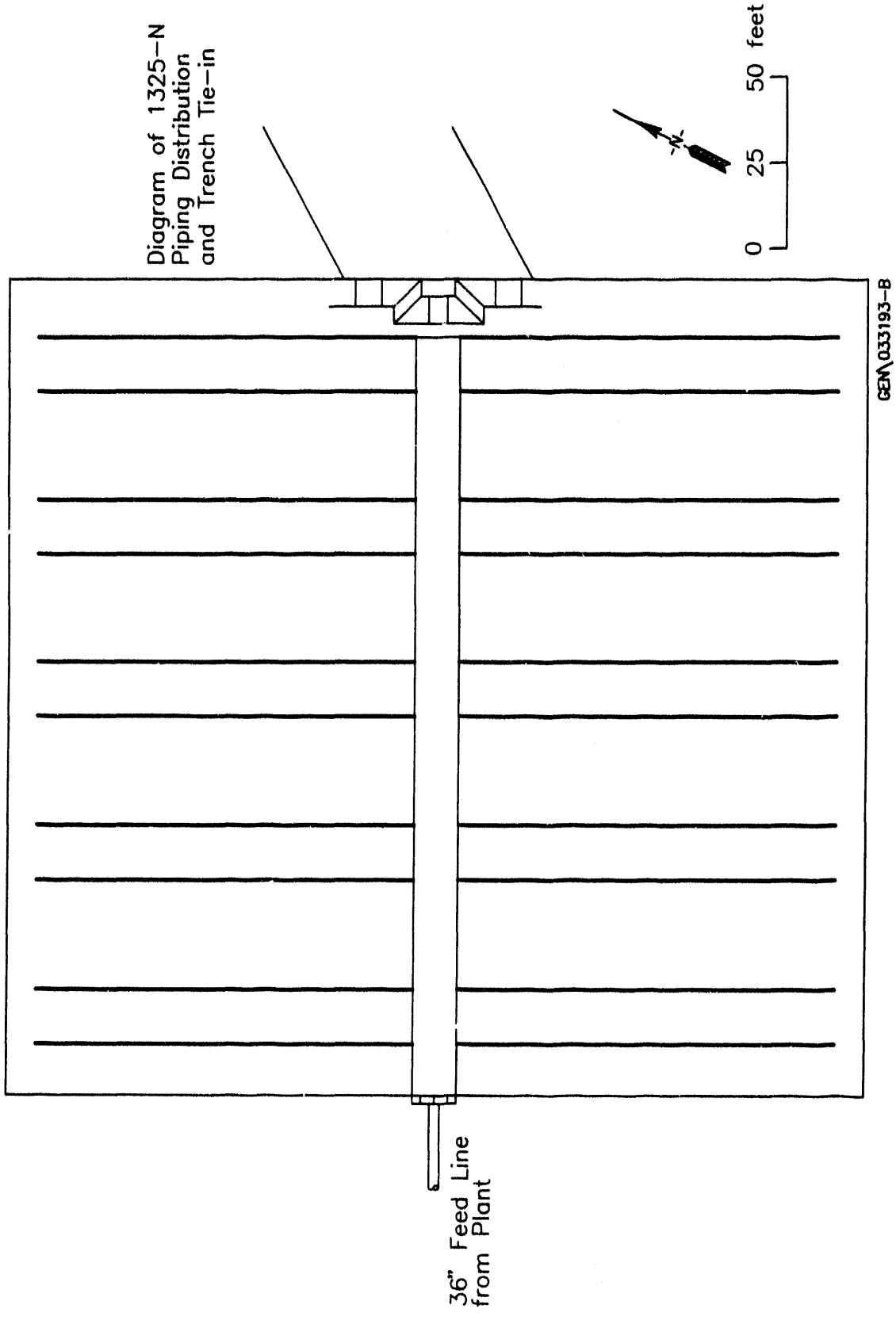


Figure 3. Plan View of the 1325-N Liquid Waste Disposal Facility.

Figure 4. Plan View of the Crib Section of the 1325-N Liquid Waste Disposal Facility.



2.3.2 1301-N LWDF

The 1301-N LWDF also is a crib and trench facility (see Figure 2) that is constructed within the boundaries of the 100-N Area perimeter fence. The crib is 88 by 38 by 4 m (290 by 125 by 12 ft), and is filled with a 1-m (3-ft) layer of boulders. Precast concrete cover panels were placed over the trench in 1982 to minimize both intrusion by wildlife and airborne contamination from the facility (DOE-RL 1991a). The zig-zag extension trench is 488 by 15 by 4 m (1,600 by 50 by 12 ft). The trench was added to the crib in 1965 because wastewater volume exceeded the capacity of the crib. The 1301-N LWDF began operation in 1965 and was used until September 1985 (DOE-RL 1991a).

2.3.3 1324-N/NA Surface Impoundment/Percolation Pond

2.3.3.1 1324-NA Percolation Pond (1324-NA Pond). The 1324-NA Pond is a large unlined percolation pond located approximately 914 m (3,000 ft) from the 105-N Building (see Figure 2). The pond was placed in service in August 1977, and was used to treat corrosive regeneration effluent from the 163-N Demineralization Plant and filter backwash water from the 183-N Filtered Water Plant. The corrosive effluent was treated in the pond by alternating the addition of acidic cation column regeneration effluent and alkaline anion column regeneration effluent, which neutralized the solutions. The pond also was to make use of the natural buffering capacity and calcareous nature of the soil underlying the pond to neutralize the waste (DOE-RL 1991a). However, no analytical data exist to verify the exact buffering capacity of the 100-N soil column.

In the spring of 1983, the 1324-NA Pond was enlarged from a bottom area of 855 m^2 (9,200 ft^2) with a volume of approximately 4,542,480 L (1,200,000 gal) to a bottom area of 2,694 m^2 (29,000 ft^2) with a volume of 11,356,200 L (3,000,000 gal). The filter backwash was routed to the Backwash Lake disposal site at this time (see Figure 2). Use of the 1324-NA Pond to treat dangerous wastes was discontinued in May 1986, when the 1324-N Surface Impoundment was put into service to treat the corrosive effluents (DOE-RL 1991a).

This facility was used from 1986 to 1988, when it was replaced by an elementary neutralization unit located at the 163-N Demineralization Plant. From 1986 until present, the pond has received the neutralized regeneration effluent from the elementary neutralization unit (DOE-RL 1991a).

2.3.3.2 1324-N Surface Impoundment (1324-N). The 1324-N Facility is a double-lined pond with leachate collection and leak detection systems, located 914 m (3,000 ft) southeast of the 105-N Building. The impoundment is approximately 43 by 23 m (140 by 75 ft) at grade and slopes to 24 by 5 m (80 by 15 ft) below grade (5 m [15 ft]) and is designed to hold 1,605,001 L (424,000 gal) (UNC 1987). The 1324-N Facility was used to treat and neutralize corrosive wastes before they were discharged to the 1324-NA Pond (UNC 1987). The 1324-N Surface Impoundment was constructed in 1986 and used until 1988. No leaks have been detected from this site.

2.3.4 105-N Reactor

The 105-N Reactor Building housed a graphite-moderated, light water-cooled, horizontal-pressure tube nuclear reactor. The reactor coolant circulating pumps were single-stage, horizontal centrifugal pumps with high-pressure water injection seals to prevent reactor coolant loss (WHC 1989a). The reactor was designed to operate in two modes: (1) production of special nuclear material only, and (2) production of special nuclear material and byproduct steam (which was used by the Hanford Generating Plant to produce electricity). When the reactor was producing special nuclear material only, steam from the secondary side of 10 steam generators was routed through 16 river water-cooled dump condensers. Condensate from these condensers was routed back into the steam generators for regeneration. Under dual-purpose operation, the steam was supplied to the Hanford Generating Plant to produce 860 MW of electrical power (WHC 1989a).

2.3.5 109-N Heat Exchanger

The portion of the reactor coolant system within the 105-N Building consists of 16 parallel lines that conducted cooling water from an inlet water manifold in the 109-N Heat Exchanger building to the reactor. Each of these 16 lines ended in a vertical header to which 54 to 66 individual pressure tube header-to-inlet nozzle connectors were attached. Similar outlet risers and parallel lines conducted the coolant from the pressure tube outlet nozzle-to-header connectors to an outlet water manifold (WHC 1989a).

In the 109-N Building, which is immediately adjacent to the 105-N Building, the reactor coolant system consisted of six cells in parallel, each containing two steam generators also in parallel, a circulating pump and associated valves, and instrumentation. Piping and steam generators in each of the six cells could be isolated from the main header piping by means of isolation valves (WHC 1989a).

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3.0 1325-N LWDF EFFLUENT CHARACTERISTICS

3.1 SOURCES

The source of effluent to the 1325-N LWDF was the 105-N and 109-N Facilities. The waste streams changed over time, because of the shutdown of some of the operations in the 100-N Area. There were five major waste streams from these facilities routed to the 1325-N LWDF:

- Reactor coolant system bleedoff
- Spent fuel storage basin cooling water overflow
- Reactor periphery cooling systems bleedoff
- Reactor primary coolant loop decontamination rinse solution
- Building drains containing radioactive wastes generated from reactor support facilities.

Table 1 is a listing of the various effluent stream sources and types for the 1325-N LWDF.

3.2 SYSTEM CONFIGURATION AND OPERATION

The 1325-N LWDF received effluent from the facilities listed in Table 2; the facilities' physical layout is shown in Figure 5. The facilities described above are connected by underground piping. The N Reactor effluent discharged to the 1325-N LWDF through the 91.4-cm- (36-in.-) diameter low-pressure flush line (FL[LP]) (Figure 6). Effluent sources (Figure 7), including various floor drains and the fuel basin overflow weirs, drained into the lift station pump well on the west side of the 105-N Facility. From the lift station, the wastewater was pumped to the 1325-N LWDF through the 91.4-cm- (36-in.-) diameter FL(LP). The emergency dump tank discharged into the 91.4-cm- (36-in.-) diameter FL(LP), just north of the lift station, via the 76.2-cm- (30-in.-) diameter overflow drain (Figure 8) (WHC 1990c).

Contaminated wastewater exited the east end of the 109-N Facility via the 15.2-cm- (6-in.-) diameter radioactive drain (RDR) and was routed to the 1310-N Facility (Figure 9). This line was designed to carry nuclear sump pump discharge and liquid waste from cell decontamination in the 109-N Facility. Currently the line carries intermittent flows of water collected from shop areas and air conditioning units within the facility (WHC 1990c).

The 25.4-cm- (10-in.-) diameter RDR routes wastewater from the west end of the 109-N Facility (see Figure 8), around the west end of the 105-N Facility, and through the 1909-N Valve Pit. Just east of the 1909-N Valve Pit, the 25.4-cm- (10-in.-) diameter RDR increases to 30.5 cm (12 in.) in diameter and runs alongside the 91.4-cm- (36-in.-) diameter FL(LP) until it reaches the 1301-N Weir Box. The stream carried by the 30.5-cm- (12- in.-) diameter RDR can be diverted to the 1310-N Facility at the

Table 1. Past and Present Effluent Stream Sources and Amounts Discharged to the 1325-N Liquid Waste Disposal Facility (adapted and modified from Diediker and Hall [1987]). (2 sheets)

Waste stream origin	Release period	Effluent/frequency	Frequency	Amounts	
				Peak reactor operations	Present limit
Reactor coolant system bleedoff	1983-1990	• Demineralized water treated with ammonium hydroxide and hydrazine	Continuous	757 L/min (200 gal/min) bleedoff and leakage	Less than or equal to 8 L/min (2 gal/min) averaged over the calendar month; not to exceed 6.8 million L (1.8 million gal) total to June 1995, all sources
Spent fuel storage basin cooling water overflow	1983-1991	• Filtered water with chlorine added as an algicide	Contingency, overflow	Varied	
Reactor periphery cooling systems bleedoff			Continuous, all sources	Bleedoff and spillage, all sources	
• Graphite and shield cooling	1983-1990	• Demineralized water treated with ammonium hydroxide and hydrazine			
• Reactor control rod cooling	1983-1990	• Demineralized water treated with ammonium hydroxide and hydrazine			
• Reactor secondary coolant loop	1983-1990	• Demineralized water treated with morpholine and hydrazine			
Reactor primary coolant loop decontamination rinse solution	1983-1987	• TURCO[a] (70% phosphoric acid), diethylthiourea, and demineralized water	Once every 2 to 4 years	Decon solution: (1) 79,493 L (21,000 gal) TURCO,[a] (2) 136 to 181 kg (300 to 400 lb) diethylthiourea; diluted to 8 wt% phosphoric acid using demineralized water	

Table 1. Past and Present Effluent Stream Sources and Amounts Discharged to the 1325-N Liquid Waste Disposal Facility (adapted and modified from Diediker and Hall [1987]). (2 sheets)

Waste stream origin	Release period	Effluent/frequency	Frequency	Amounts	
				Peak reactor operations	Present limit
Building drains containing radioactive wastes generated from reactor support facilities					
• Laboratory wastes	1983-1986	• Chemical analyses	• Batch mode, varied	• 9,854 L/yr (2,600 gal/yr)	• N/A
• Auxiliary power battery lockers	1983-1987	• Waste from nickel-cadmium and lead-acetate batteries	• Unknown, only suspected source	• 302 to 379 L/yr (80 to 120 gal/yr)	• N/A
• Hydrazine mixing and injecting area	1983-1986	• Hydrazine	• Unknown, only suspected source	• 158 kg/yr (350 lb/yr)	• N/A

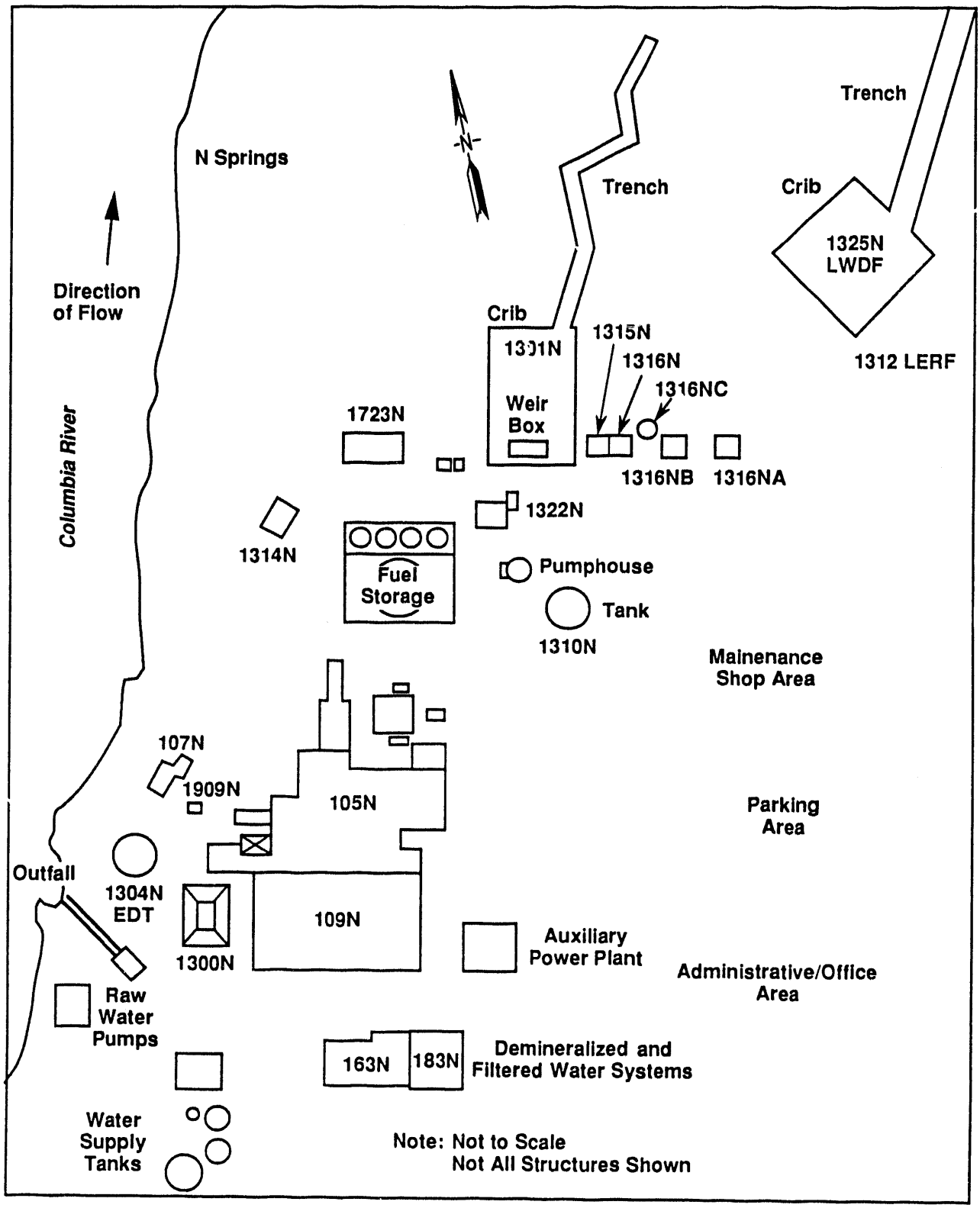
[a]TURCO is a trademark of Turco Products, Incorporated.
N/A = not available.

Table 2. Facilities that Drain to the 1325-N Liquid Waste Disposal Facility (WHC 1990c).

Facility	Facility description	Connection to the 1325-N LWDF (cm in diameter [in. in diameter])
105-N	Reactor Facility	91.4 cm (36 in.) FL(LP)
109-N	Heat Exchanger Building	25.4 cm (10 in.); RDR 30.5 cm (12-in.)
1301-N	Liquid Waste Disposal Facility and Weir Box	91.4 cm (36 in.) FL(LP)
1304-N	Emergency Dump Tank	91.4 cm (36 in.); Overflow to 91.4 cm (36 in.) FL(LP)
1310-N	Radioactive Chemical Waste Treatment Facility	30.5 cm (12 in.); RDR 61 cm (24 in.)
1315-N	Reactor Effluent Diversion System Valve House	91.4 cm (36 in.) FL(LP)
1316-N	Valve House	91.4 cm (36 in.) FL(LP)
1316-NA	Valve Vault Building	91.4 cm (36 in.) FL(LP)
1316-NB	Magnetic Flow Meter Vault	91.4 cm (36 in.) FL(LP)
1316-NC	Turbine Flow Meter Vault	91.4 cm (36 in.) FL(LP)
1322-N	Waste Treatment Pilot Plant and Valve Pit	91.4 cm (36 in.) FL(LP); RDR 30.5 cm (12 in.)
1322-NA	Effluent Water Pilot Plant	91.4 cm (36 in.) FL(LP); RDR 30.5 cm (12 in.)
1322-NB	Crib Effluent Iodine Monitoring Facility	91.4 cm (36 in.) FL(LP); RDR 30.5 cm (12 in.)
1322-NC	Crib Sample Pump Pit	91.4 cm (36 in.) FL(LP); RDR 30.5 cm (12 in.)
1327-N	Valve House	91.4 cm (36 in.) FL(LP)
1723-N	Materials Receiving, Inspection, and Storage Building	10.2 cm (4 in.) cold drain
1909-N	Waste Disposal Valve Pit	25.4 cm (10 in.); RDR 30.5 cm (12 in.)

FL(LP) = low-pressure flush line.
RDR = radioactive drain.

Figure 5. Layout of the 100-N Area (WHC 1990c).



H9309026.8

Figure 6. Route of N Reactor Effluent to the 1325-N Liquid Waste Disposal Facility via the 91.4-cm- (36-in.-) Diameter Low-Pressure Flush Line (WHC 1990c).

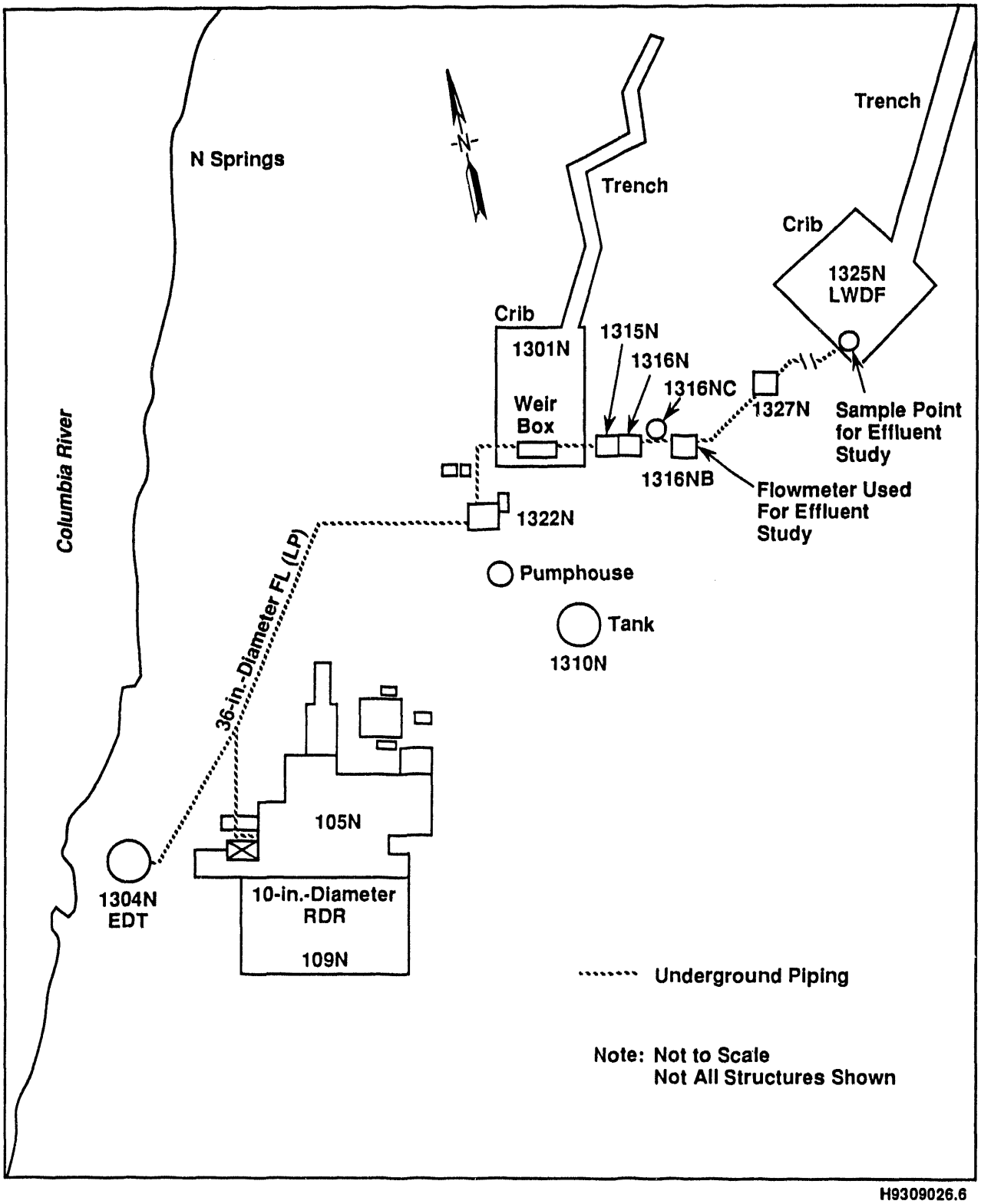


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Figure 7. Schematic of N Reactor Effluent Discharged to the 1325-N Liquid Waste Disposal Facility (WHC 1990c).

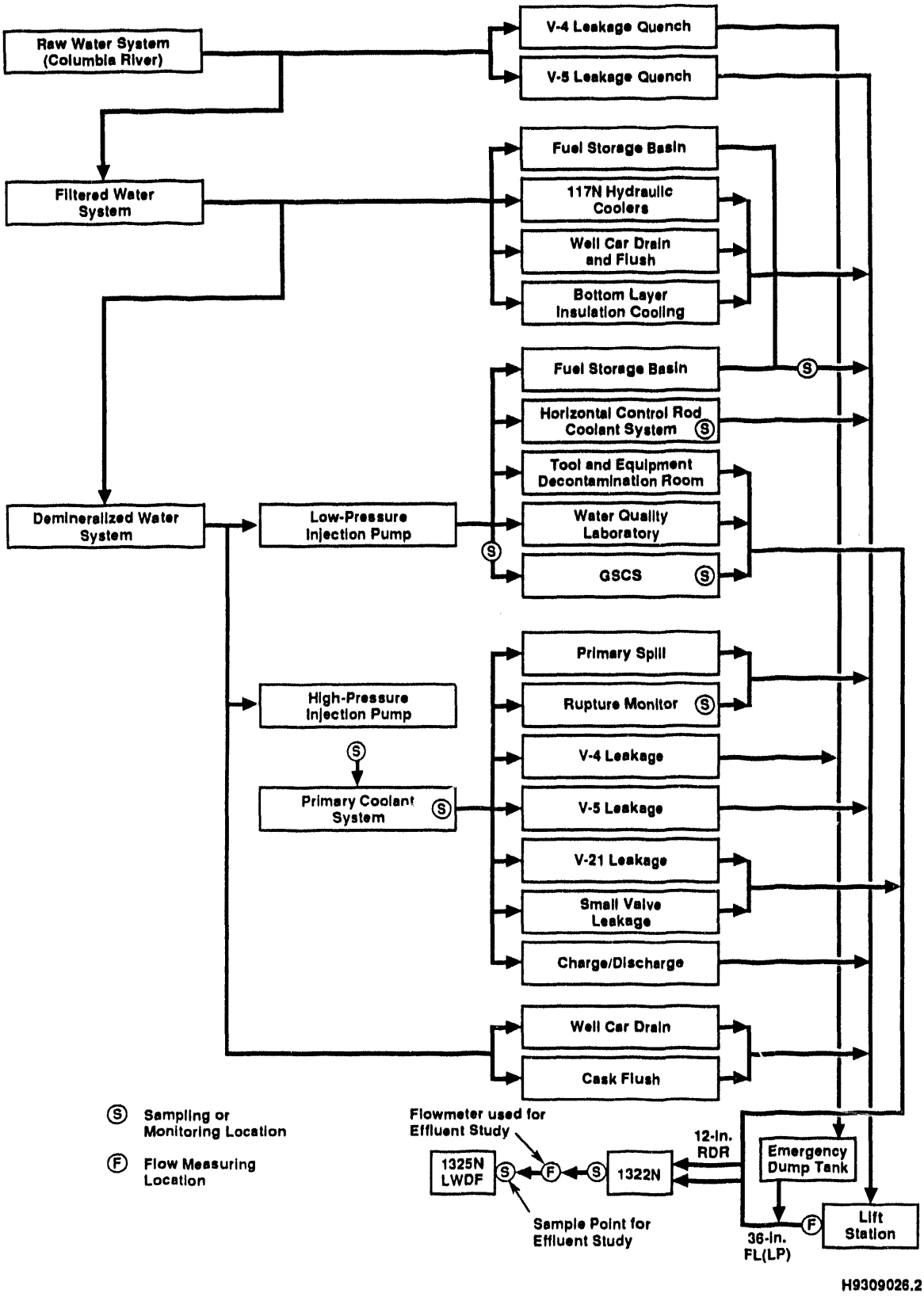


Figure 8. Effluent Discharge Routes from the 105-N Reactor Building and Adjacent Facilities (WHC 1990c).

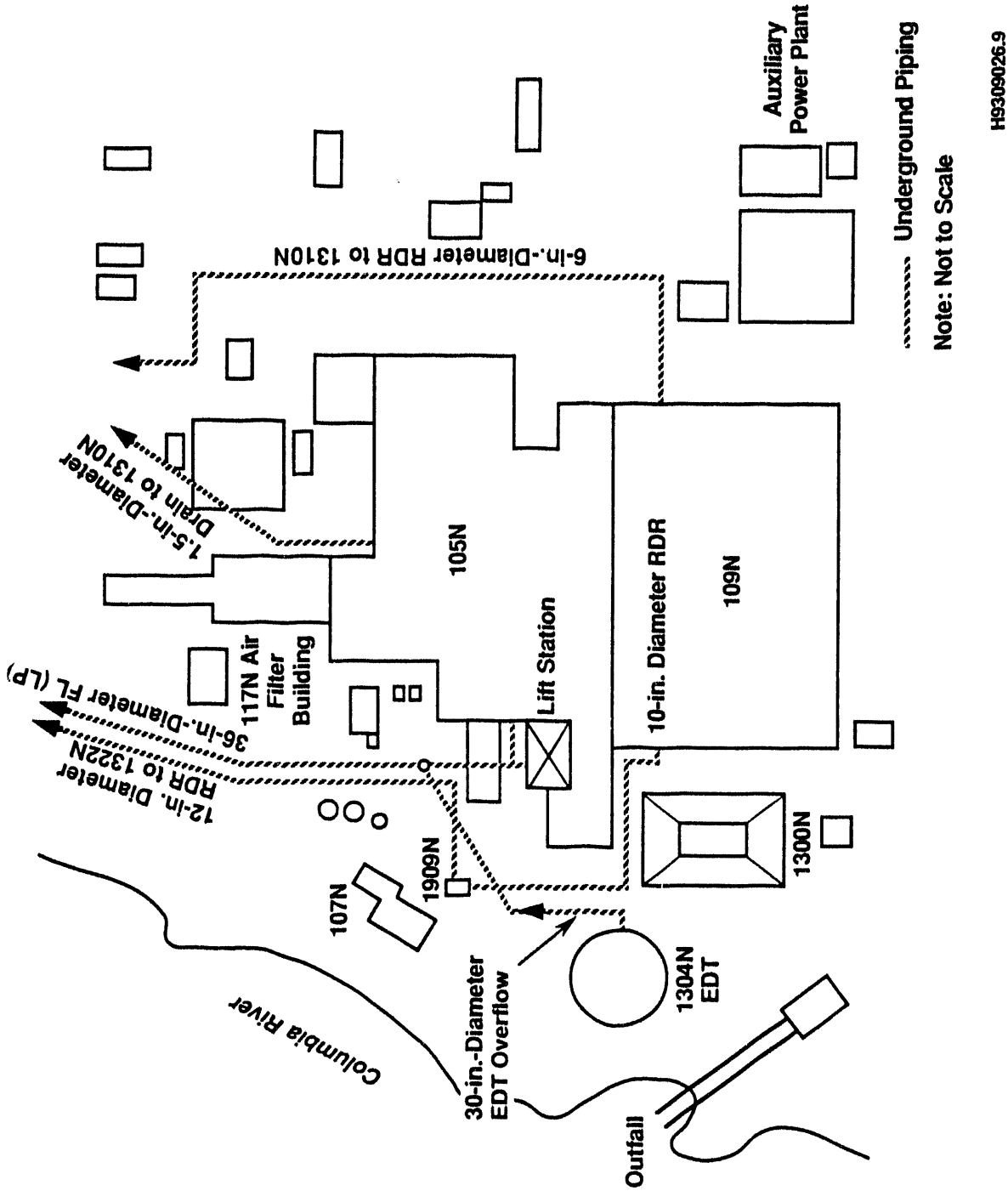
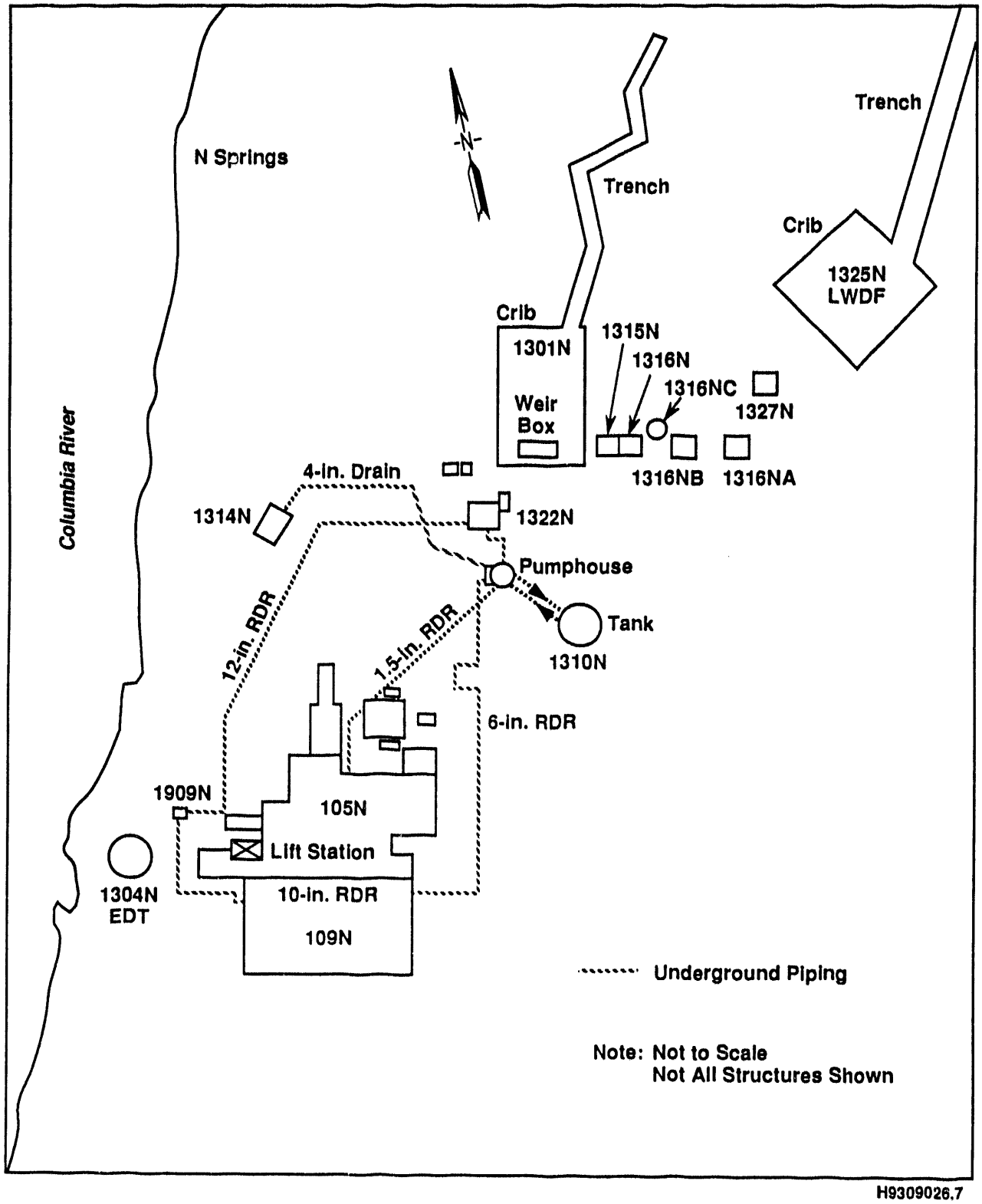


Figure 9. Effluent Discharge Routes to the 1314-N Liquid Waste Railcar Loadout Station During Various Decontamination Activities (WHC 1990c).



1322-N Facility. This option was used during a major plant decontamination (see Figure 9). This radioactive drain currently carries a small, continuous flow from cell sumps in the 109-N Facility (WHC 1990c).

The 10.2-cm- (4-in.-) diameter chemical waste line discharged from the 1310-N Facility to the 1314-N Loadout Facility. This provided a means of disposing of liquid wastes generated during decontamination activities, which were more concentrated or corrosive than the usual N Reactor effluent discharged to 1325-N LWDF (WHC 1990c).

3.3 DISCHARGE VOLUME AND FLOW REGIME

During peak reactor operations the flow rate out to the 1325-N LWDF ranged from 3,785 to 6,057 L/min (1000 to 1,600 gal/min) and could increase to 7,570 L/min (2,000 gal/min) during specific operations. Since April 1991, there has been no discharge to the facility. There are no plans to resume discharge to the facility on a regular basis and all discharges to the facility are required to be terminated by June 1995 (in accordance with Tri-Party Agreement Milestone M-17-15A). However, the facility may need to be used on an emergency basis before that time, for discharge of fire protection water. If there is further discharge to the facility, the following limits are in effect in accordance with Tri-Party Agreement Milestone M-17-15A:

- Flow rate must be less than or equal to 7.6 L/min (2 gal/min) averaged over the calendar month
- Maximum discharge amount allowed to the facility from the present time until June 1995 is 6,813,741 L (1,800,000 gal).

The maximum amount of effluent that could be discharged in an average month would be:

$$30 \text{ days} \times 1{,}440 \text{ min/day} = 43{,}200 \text{ minutes} \quad (1)$$

$$(43{,}200 \text{ minutes} \times 7.6 \text{ L/min} = 328{,}320 \text{ L})$$
$$43{,}200 \text{ minutes} \times 2 \text{ gal/min} = 86{,}400 \text{ gal.} \quad (2)$$

Thus, the maximum amount that could be discharged in a given month is 328,320 L (86,400 gal).

3.4 EFFLUENT CONSTITUENTS

3.4.1 Quantities

The total effluent discharged to the 1325-N LWDF from January 1983 to April 1991 is 7,305,844,743 L (1,930,000,000 gal). Table 1 explains these numbers in more detail.

3.4.2 Loading Rates

The information in this section was adapted from the *Closure/Post-Closure Plan for the 1301-N and 1325-N Liquid Waste Disposal Facilities* (Diediker and Hall 1987) and the *Liquid Effluent Study Final Project Report* (WHC 1990a). This section contains actual and potential waste sources for the 1325-N LWDF treatment, storage, and disposal unit as well as radiological and chemical loading estimates (Table 3).

3.4.2.1 Estimate of Nondangerous Waste. This section is summarized from Diediker and Hall (1987). The major portion of the effluent discharged to the 1325-N LWDF (from 1983 to April 1991) has come from the 105-N/109-N Facilities. The five identified waste streams that emanate from these facilities were sampled at the point of discharge into the 1325-N LWDF (Table 4). The results indicated that the effluent did not exhibit any of the characteristics of a dangerous waste. None of the results met the criteria to be designated dangerous wastes, as listed in *Washington Administrative Code* (WAC) 173-303-090.

3.4.2.2 Estimate of Potential Dangerous Waste. This section is summarized from Diediker and Hall (1987) and addresses the potential for dangerous wastes to have been discharged to the 1325-N LWDF. The following potentially dangerous waste constituents were identified using process knowledge and historical information: (1) ammonium hydroxide, (2) hydrazine, (3) chlorine, (4) morpholine, (5) TURCO 4512-A[1] (70% phosphoric acid), (6) chemical laboratory wastes, and (7) nickel-cadmium and lead-acetate batteries. The estimated maximum potential volume of dangerous wastes received by the 1325-N LWDF is given in Table 1. Each of these constituents will be discussed in the following sections as part of the waste stream that produces it.

3.4.2.2.1 Reactor Primary Coolant System. The reactor primary coolant system was supplied by demineralized water with chemicals added for water quality control. The chemicals that were introduced into the primary coolant system were ammonium hydroxide and hydrazine. Ammonium hydroxide was used for pH control and was injected at a concentration of approximately 40 parts per million (ppm) to maintain a pH of 10.2 to 10.4. Hydrazine was introduced for oxygen control at a concentration of 0.04 ppm. Normal operation of the reactor primary coolant system resulted in approximately 757 L/min (200 gal/min) of bleedoff and leakage that was discharged to the 1325-N LWDF. Concentration of chemicals used in maintaining the water quality of the primary coolant system was very low at the point of discharge to the 1325-N LWDF. Their influence was nondetectable and the waste stream did not exhibit any of the characteristics of a dangerous waste because of these chemicals.

3.4.2.2.2 Fuel Storage Basin Cooling Water. The spent fuel storage was supplied by filtered water treated with chlorine (as an algicide). A trace amount of residual chlorine was maintained to ensure complete treatment. The overflow did not contain hazardous chemical constituents and therefore had no impact on the dangerous waste designation of wastes discharged to the

[1]TURCO 4512-A is a trademark of Turco Products, Incorporated.

Table 3. Radionuclide and Chemical Loading, N Reactor Effluent (WHC 1990a).

Flow rate: 4.95 E+07 L/month					
Constituent	kg/L	kg/month	Constituent	kg/L	kg/month
Barium	3.67 E-08	1.82 E+00	Unknown glycol	2.80 E-08	1.39 E+00
Boron	6.12 E-08	3.03 E+00	Unknown hydrocarbon	1.90 E-07	9.41 E+00
Cadmium	2.00 E-09	9.90 E-02	Unknown oxygenated PAH	9.00 E-09	4.45 E-01
Calcium	1.16 E-05	5.74 E+02	Unknown phthalate	9.00 E-09	4.45 E-01
Chloride	1.13 E-06	5.59 E+01	Alpha activity[a]	2.94 E-11	1.46 E-03
Fluoride	1.48 E-07	7.33 E+00	Beta activity[a]	7.03 E-09	3.48 E+00
Iron	2.53 E-07	1.25 E+01	Total dissolved solids	3.97 E-05	1.97 E+03
Magnesium	2.90 E-06	1.44 E+02	Total organic carbon	7.25 E-07	3.59 E+01
Manganese	6.75 E-09	3.34 E+01	Total carbon	8.53 E-06	4.22 E+02
Nitrate	4.02 E-07	1.99 E+01	TOX (as chloride)	1.80 E-08	8.91 E-01
Potassium	4.75 E-07	2.35 E+01	Americium-241[a]	3.48 E-11	1.72 E-03
Silicon	1.48 E-06	7.33 E+01	Curium-242[a]	1.67 E-13	8.27 E-06
Sodium	1.53 E-06	7.57 E+01	Curium-244[a]	3.68 E-13	1.82 E-05
Strontium	5.95 E-08	2.95 E+00	Cobalt-60[a]	4.36 E-10	2.16 E-02
Sulfate	1.06 E-05	5.25 E+02	Cesium-134[a]	9.43 E-11	4.67 E-03
Uranium	1.96 E-09	9.70 E-02	Cesium-137[a]	2.29 E-08	1.13 E+00
Zinc	1.07 E-08	5.30 E-01	Carbon-14[a]	2.22 E-12	1.10 E-04
Acetone	9.78 E-09	4.84 E-01	Hydrogen-3 (tritium)[a]	1.02 E-07	5.05 E-00
Ammonia	1.82 E-07	9.01 E+00	Manganese-54[a]	3.19 E-10	1.58 E-02
Benzoic acid	1.10 E-07	5.45 E+00	Lead-210[a]	1.22 E-12	6.04 E-05
1-butanol	2.70 E-08	1.34 E+00	Plutonium-238[a]	4.82 E-12	2.39 E-04
2-butanone	1.72 E-08	8.51 E-01	Plutonium-239/240[a]	3.62 E-11	1.79 E-03
Butylated hydroxy toluene	1.07 E-08	5.30 E-01	Radium-total[a]	1.78 E-13	8.81 E-06
Hydrazine	3.05 E-08	1.51 E+00	Ruthenium-106[a]	4.63 E-10	2.29 E-02
MIBK (hexone)	1.01 E-08	5.00 E-01	Strontium-90[a]	8.40 E-08	4.16 E+00
Toluene	5.08 E-09	2.51 E-01	Uranium-234[a]	1.34 E-12	6.63 E-05
Trichloromethane	4.85 E-09	2.40 E-01	Uranium-235[a]	1.51 E-13	7.47 E-06
Unknown	4.70 E-08	2.33 E+00	Uranium-238[a]	9.12 E-13	4.51 E-05

Notes: (1) Data collected from October 1989 through March 1990.
(2) Flow rate is the average of rates from the N Reactor Effluent Stream-Specific Report (WHC 1990c, Addendum 3).
(3) Constituent concentrations are average values from the statistics in the N Reactor Effluent Stream-Specific Report (WHC 1990c, Addendum 3).

[a]Concentration units of these constituents are reported as curies per liter. Loading units of these constituents are reported as curies per month.

MIBK = methyl isobutyl ketone.
PAH = polycyclic aromatic hydrocarbon.
TOX = total organic halogen.

Table 4. 1325-N Liquid Waste Disposal Facility Analysis (Diediker and Hall 1987). (2 sheets)

Parameter (MDL)	Sample			
	1	2	3	Average
pH (standard units)	6.58	6.56	6.97	6.70
Conductivity (µmhos/cm)	148	155	190	164
Mercury (0.001 ppm)	LD	LD	LD	LD
Ethylene glycol (10 ppm)	LD	LD	LD	LD
Enhanced thiourea (0.2 ppm)	LD	LD	LD	LD
TOC (1 ppm)	0.00184	0.00200	0.00205	0.00197
Cyanide (0.01 ppm)	LD	LD	LD	LD
Barium (0.006 ppm)	0.030	0.027	0.027	0.028
Cadmium (0.002 ppm)	LD	LD	LD	LD
Chromium (0.01 ppm)	LD	LD	LD	LD
Lead (0.03 ppm)	LD	LD	LD	LD
Silver (0.01 ppm)	LD	LD	LD	LD
Sodium (0.1 ppm)	1.831	1.819	1.781	1.810
Nickel (0.01 ppm)	LD	LD	LD	LD
Copper (0.01 ppm)	LD	LD	LD	LD
Vanadium (0.005 ppm)	LD	LD	LD	LD
Antimony (0.1 ppm)	LD	LD	LD	LD
Aluminum (0.15 ppm)	LD	LD	LD	LD
Manganese (0.005 ppm)	LD	LD	LD	LD
Potassium (0.1 ppm)	0.647	0.608	0.606	0.620
Iron (0.05 ppm)	0.081	0.077	0.050	0.069
Beryllium (0.005 ppm)	LD	LD	LD	LD
Osmium (0.3 ppm)	LD	LD	LD	LD
Strontium (0.3 ppm)	LD	LD	LD	LD
Zinc (0.005 ppm)	LD	LD	LD	LD
Calcium (0.05 ppm)	14.400	13.970	14.050	14.140
Nitrate (0.5 ppm)	LD	LD	LD	LD
Sulfate (0.5 ppm)	12.416	11.532	11.970	11.973
Fluoride (0.5 ppm)	LD	LD	LD	LD
Chloride (0.5 ppm)	1.578	1.478	1.533	1.530
Phosphate (1 ppm)	LD	LD	LD	LD
Phosphorus pesticides (0.005 ppm)	LD	LD	LD	LD
Chlorinated pesticides (0.001 ppm)	LD	LD	LD	LD
Enhanced ABN list	LD	LD	LD	LD
Citrus red (1 ppm)	LD	LD	LD	LD
Arsenic (0.005 ppm)	LD	LD	LD	LD
Ammonium ion (0.05 ppm)	LD	LD	LD	LD

Table 4. 1325-N Liquid Waste Disposal Facility Analysis (Diediker and Hall 1987). (2 sheets)

Parameter (MDL)	Sample			
	1	2	3	Average
Coliform (3 MPN)	--	0.023	0.009	0.016
Selenium (0.005 ppm)	LD	LD	LD	LD
Thallium (0.01 ppm)	LD	LD	LD	LD

ABN = acid-base neutrals.
LD = less than detectable.
MDL = minimum detection limit.
MPN = most probable number.
TOC = total organic carbon.
Note: Data obtained from samples taken August 1985.

1325-N LWDF. Since December 1985, the spent fuel storage basin was supplied by demineralized water that does not contain chlorine.

3.4.2.2.3 Reactor Periphery Cooling Systems. Reactor periphery cooling systems that discharge bleedoff wastes to the 1325-N LWDF include the following:

- Graphite and shield cooling system--supplied by demineralized water with chemicals added for water quality control. Ammonium hydroxide was injected at a concentration of approximately 40 ppm to maintain a pH of 10.0 to 10.2. Hydrazine was injected for oxygen control at a concentration of 0.04 ppm.

- Reactor control rod cooling system--recirculating system supplied by demineralized water with chemicals added for water quality control. Ammonium hydroxide was injected at a concentration of approximately 40 ppm to maintain a pH of 7.0. Hydrazine was added for oxygen control at a concentration of 0.15 ppm.

- Reactor secondary coolant loop--supplied by demineralized water with chemicals added for water quality control. Morpholine was injected at a concentration of approximately 4 ppm to maintain a pH of 8.6 to 9.2. Hydrazine was injected for oxygen control at a concentration of no greater than 1 ppm.

As with other reactor cooling systems, bleedoff and spillage from the periphery cooling systems resulted in small continuous discharges to the 1325-N LWDF. Sampling conducted on the 1325-N LWDF influent (see Table 4) indicates that the flow from these streams did not change the dangerous waste designation of wastes discharged to the 1325-N LWDF.

3.4.2.2.4 Reactor Primary Coolant Loop Decontamination. A decontamination of the reactor primary coolant loop was performed once every 2 to 4 years, as necessary. The decontamination solution is made up of 79,494 L (21,000 gal) of TURCO 4512-A (70% phosphoric acid) and 136 to 181 kg (300 to 400 lb) of diethylthiourea. This solution was diluted to an 8 wt% phosphoric acid solution as it entered the reactor coolant loop.

The decontamination solution and concentrate rinse were disposed of through a system separate from the 1325-N LWDF, using a radioactive waste tank. After the pH of the rinsate had been verified between 6.0 and 9.0, the final rinse solution, containing approximately 378,541 L (100,000 gal) of demineralized water, was discharged to the 1325-N LWDF.

Wastes received at the 1325-N LWDF from the reactor decontamination were extremely dilute (approximately one ten-thousandth of the original concentration of the decontaminated wastes). The calculated amount of phosphoric acid solution that was released to the 1325-N LWDF is 5.7 L (1.5 gal) per decontamination. The calculated amount of diethylthiourea released per decontamination was 2.3 g (0.08 oz). The decontamination rinse solution was not designated as a dangerous waste because the concentrations of the component chemicals were extremely low when the rinsate was diverted to the 1325-N LWDF.

3.4.2.2.5 Building Drains. The radioactive drain system was a network of floor drains that collected radioactive water from throughout the 109-N and 105-N Facilities. Pump leakage and system bleedoff from the reactor primary and periphery cooling systems were transported to the 1325-N Facility via this system. Other contributing streams to this drain system were generated by laboratories, decontamination activities, and other routine functions at the N Reactor. Of the wastes discarded to the radioactive drain system, three have exhibited characteristics of a dangerous waste.

Leaks and spills from the auxiliary power battery lockers have contributed 303 to 454 L/yr (80 to 120 gal/yr) of waste from nickel-cadmium and lead-acetate batteries. It is estimated that approximately 40% of the spilled material was from the nickel-cadmium batteries and 60% was from the lead-acetate batteries.

Spills of hydrazine from the hydrazine mixing and injection facilities are believed to have entered the radioactive drain system. Spills in this area were very small in volume and, in the case of the mixed solution, were extremely dilute. It is estimated that a maximum of 159 kg/yr (350 lb/yr) of hydrazine were spilled in this manner. The mixing area was curbed and isolated from the building drain in September 1986, thereby allowing spills to be contained, cleaned up, and disposed of in accordance with WAC 173-303 regulations.

Chemical analyses were performed in the laboratories to determine the amount of hydrazine, ammonia, chloride, and fluoride in the reactor coolant loop water. Waste characterization indicated that the solution discarded from these analyses (approximately 9,842 L/yr [2,600 gal/yr]) contained constituents that exhibited dangerous waste characteristics in accordance with WAC 173-303-090. Historically these solutions were discarded to the radioactive drain system, which discharges to the 1325-N LWDF. Since December 1986, all of these solutions were contained and disposed of in accordance with WAC 173-303 regulations.

Each of these wastes, at their point of introduction into the radioactive drain system, contain contaminants that are designated dangerous wastes in accordance with WAC 173-303-090. However, sampling conducted on the 1325-N LWDF influent (see Table 4) did not identify any dangerous waste characteristics at the point of discharge of the waste stream into the

1325-N LWDF. The dangerous wastes that were being discharged to the 1325-N LWDF were controlled by the following means:

- Collected and disposed of in accordance with the "Dangerous Waste Regulations" (WAC 173-303)
- Controlled through improved secondary containment
- Controlled through the application of administrative controls to prevent the discharge of dangerous waste to the 1325-N LWDF.

3.4.3 Operational Factors

The N Reactor effluent consisted primarily of cooling water and decontamination flush water used at various points within the system. Radioactive contamination of the wastewater contributors occurred during reactor operations. Sources of these radioactive constituents included activation products produced in the primary coolant system and fission products that result from occasional fuel cladding failures. Chemical contamination could occur if corrosion products and residuals remain in the system following reactor decontamination activities.

3.5 CONSTITUENTS OF INTEREST AND KEY PARAMETERS

The sample analytical data indicate that effluent discharges contain numerous radionuclides in concentrations which, in most cases, greatly exceed the WAC 173-200 Washington State Water Quality Standards (WWQS) concentration limits. Note that disposal to the crib began in 1983 but the old radiological data set included some samples taken before 1983. While concentrations of these constituents have decreased since the N Reactor was placed in dry layup, concentrations of some radioactive constituents remain high as a result of residual contamination remaining in the system. The loading data indicate that tritium, strontium-90, and cesium-137 are the principal radionuclides discharged in excess of 1 Ci/month (see Table 3).

The key constituents of interest identified in the *Liquid Effluent Study Final Project Plan* (WHC 1990a) are the long half-life radionuclides: tritium, cobalt-60, strontium-90, cesium-134, cesium-137, plutonium-238, plutonium-239, and americium-241. Of these parameters, only tritium, strontium-90, cobalt-60, and cesium-137 have exceeded the WWQS or equivalent radionuclide standards in effluent samples (WHC 1990a).

4.0 CONCEPTUAL MODEL OF HYDROLOGIC RESPONSE AND CONTAMINANT MIGRATION

4.1 HYDROGEOLOGIC FRAMEWORK

4.1.1 Regional and Hanford Site Geology

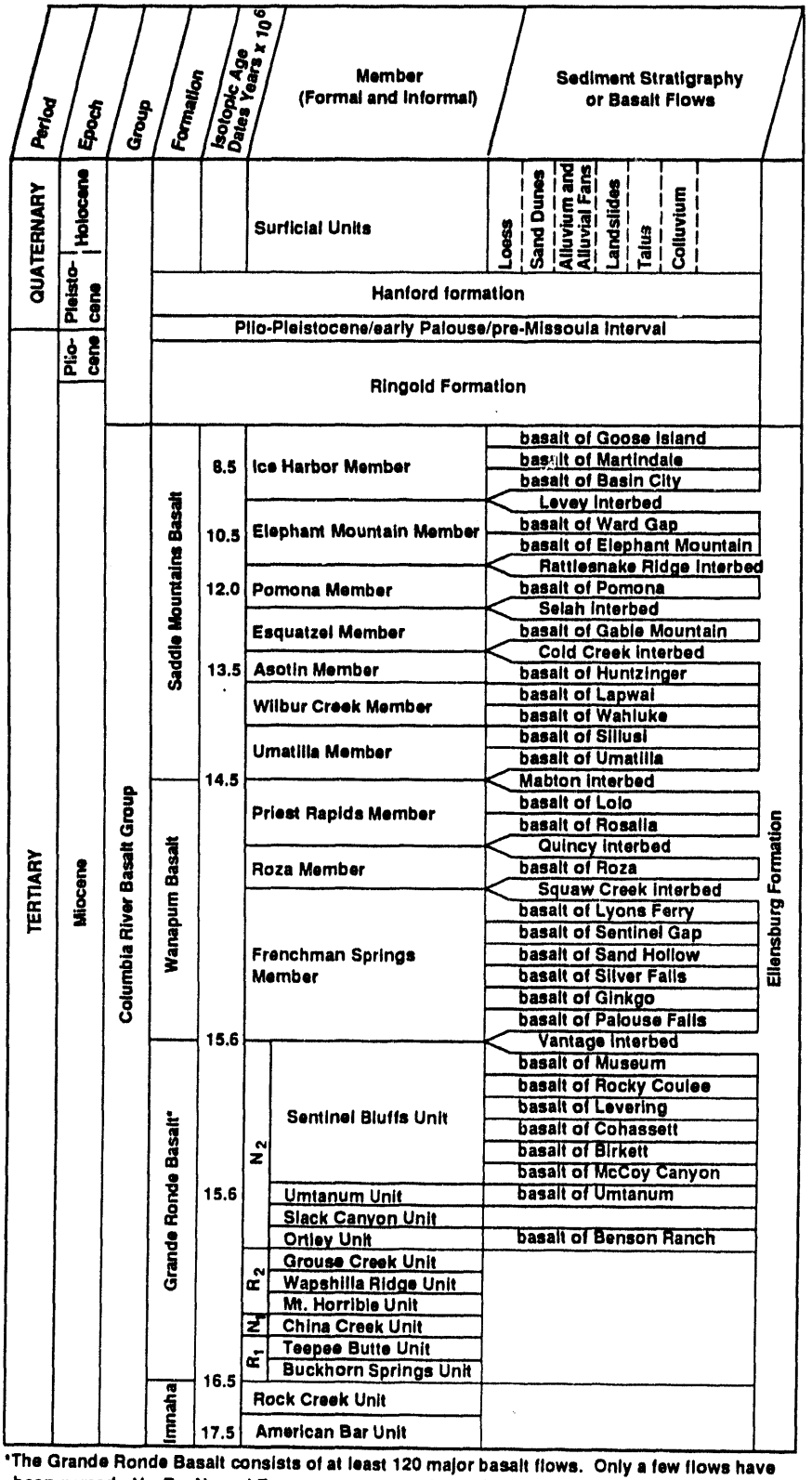
The Pasco Basin and the Hanford Site are underlain by pre-Miocene sedimentary and crystalline rocks (Campbell 1989), Miocene-aged (17.5 to 6 Ma) basalts of the Columbia River Basalt Group (CRBG) (Myers et al. 1979; Reidel and Fecht 1981; DOE 1988; Tolan et al. 1989; Reidel et al. 1989, 1992) and interbedded sediments of the Ellensburg Formation (Reidel and Fecht 1981; DOE 1988; Smith 1988), and a late-Miocene- to Holocene-aged (<8.5 Ma to present) suprabasalt sediment sequence (Myers et al. 1979; Tallman et al. 1981; DOE 1988; Smith et al. 1989; Lindsey 1991a, 1991b; Reidel et al. 1992).

4.1.1.1 Columbia River Basalt Group. The CRBG is an assemblage of tholeiitic, continental flood basalts that cover an area of more than 163,157 km^2 (63,000 mi^2) in Washington, Oregon, and Idaho, and have an estimated volume of about 174,356 km^3 (40,800 mi^3) (DOE 1988; Reidel and Hooper 1989; Tolan et al. 1989). The CRBG is divided, from oldest to youngest, into five formations: Imnaha Basalt, Picture Gorge Basalt, Grande Ronde Basalt, Wanapum Basalt, and Saddle Mountains Basalt (DOE 1988; Tolan et al. 1989) (Figure 10). The Saddle Mountains Basalt (the uppermost basalt at the Hanford Site) is divided into (from oldest to youngest) the Umatilla, Wilbur Creek, Asotin, Esquatzel, Pomona, Elephant Mountain, and Ice Harbor Members (Reidel and Fecht 1981).

4.1.1.2 Ellensburg Formation. The Ellensburg Formation consists of volcaniclastic and siliciclastic deposits that occur between CRBG basalt flows (DOE 1988; Smith 1988). At the Hanford Site the three uppermost units of the Ellensburg Formation are, from oldest to youngest, the Selah interbed, the Rattlesnake Ridge interbed, and the Levy interbed. A detailed discussion of the Ellensburg Formation at the Hanford Site is given in Reidel and Fecht (1981). Smith (1988) and Smith et al. (1989) discuss the Ellensburg Formation and correlative units throughout the region.

4.1.1.3 Suprabasalt Sediments. Discussions of various aspects of suprabasalt sediment geology are found in Myers et al. (1979); Tallman et al. (1979, 1981); PSPL (1982); Bjornstad (1984); Fecht et al. (1987); DOE (1988); Baker et al. (1989); Smith et al. (1989); Delaney et al. (1991); Lindsey (1991a, 1992); Lindsey et al. (1991); and Reidel et al. (1992). Delaney et al. (1991), Lindsey (1991a), and Reidel et al. (1992) provide the most recent synopsis of suprabasalt sediment geology for the Hanford Site. The following discussion is summarized from these recent reports.

Figure 10. Generalized Stratigraphy of the Pasco Basin and Surrounding Area.



*The Grande Ronde Basalt consists of at least 120 major basalt flows. Only a few flows have been named. N_2, R_2, N_1 and R_1 are magnetostratigraphic units.

H9102029.6b

The suprabasalt sedimentary sequence (Figure 11) is up to 229 m (750 ft) thick at the Hanford Site. It is dominated by the laterally extensive late-Miocene to Pliocene Ringold Formation and the Pleistocene Hanford formation. Laterally discontinuous units, referred to as the Plio-Pleistocene unit, early "Palouse" soil, and pre-Missoula gravels, separate the Hanford formation and Ringold Formation locally. Holocene-aged alluvial and eolian deposits cap the suprabasalt sequence.

4.1.1.3.1 Ringold Formation. The Ringold Formation is up to 183 m (600 ft) thick within the Pasco Basin. The Ringold Formation pinches out against basalt ridges around the edge of and within the basin, and it consists of semi-indurated clay, silt, fine- to coarse-grained sand, and pebble to cobble gravel. Ringold deposits are grouped into five sediment facies associations (fluvial gravel, fluvial sand, overbank-paleosol, lacustrine, basaltic alluvium) that are defined on the basis of lithology, petrology, stratification, and pedogenic alteration. The associations are summarized as follows:

(1) Fluvial gravel--Consists of clast and lesser matrix-supported pebble to cobble gravel with a fine- to medium-grained sand matrix. Grain-size distributions tend to be bimodal with granules and coarse-grained sand being rare. Crude to well-defined stratification and low-angle, lenticular bedding geometries generally dominate.

(2) Fluvial sand--Fine- to coarse-grained quartzo-feldspathic sands displaying well-defined stratification dominate. Fining upwards packages less than one to several meters thick are common.

(3) Overbank-paleosol--Laminated to massive silty sand, silt, and clay displaying evidence of pedogenic alteration dominates.

(4) Lacustrine--Characterized by well-stratified clay with interbedded silt and silty sand.

(5) Basaltic alluvium--Massive to crudely stratified, weathered to unweathered, basaltic pebble to cobble gravel, commonly with a mud-rich matrix dominates.

The distribution of facies associations within the Ringold Formation forms the basis for stratigraphic subdivision (Lindsey 1991a, 1991b). The lower half of the Ringold Formation is characterized by fluvial gravel and sand-dominated intervals designated units A, B, C, D, and E (Figure 12) that interfinger with fine-grained deposits typical of the overbank-paleosol and lacustrine facies associations. The lowest of these fine-grained intervals is designated the lower mud unit (see Figure 12). Interstratified deposits of the fluvial sand and overbank-paleosol facies associations and strata dominated by the lacustrine facies association form the upper half of the Ringold Formation (commonly referred to as the upper Ringold).

Figure 11. Generalized Stratigraphy of the Suprabasalt Sediments in the Pasco Basin.

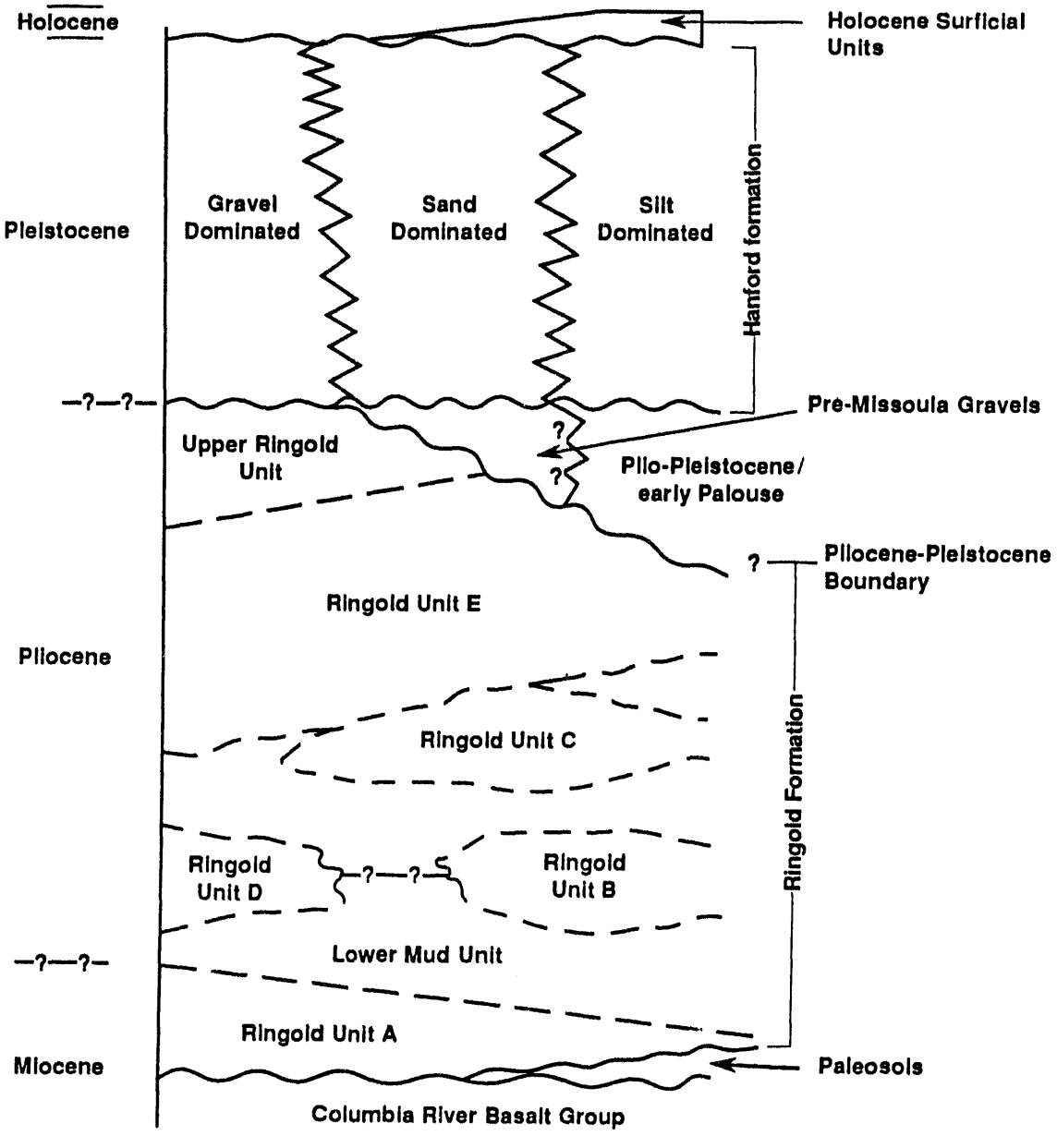
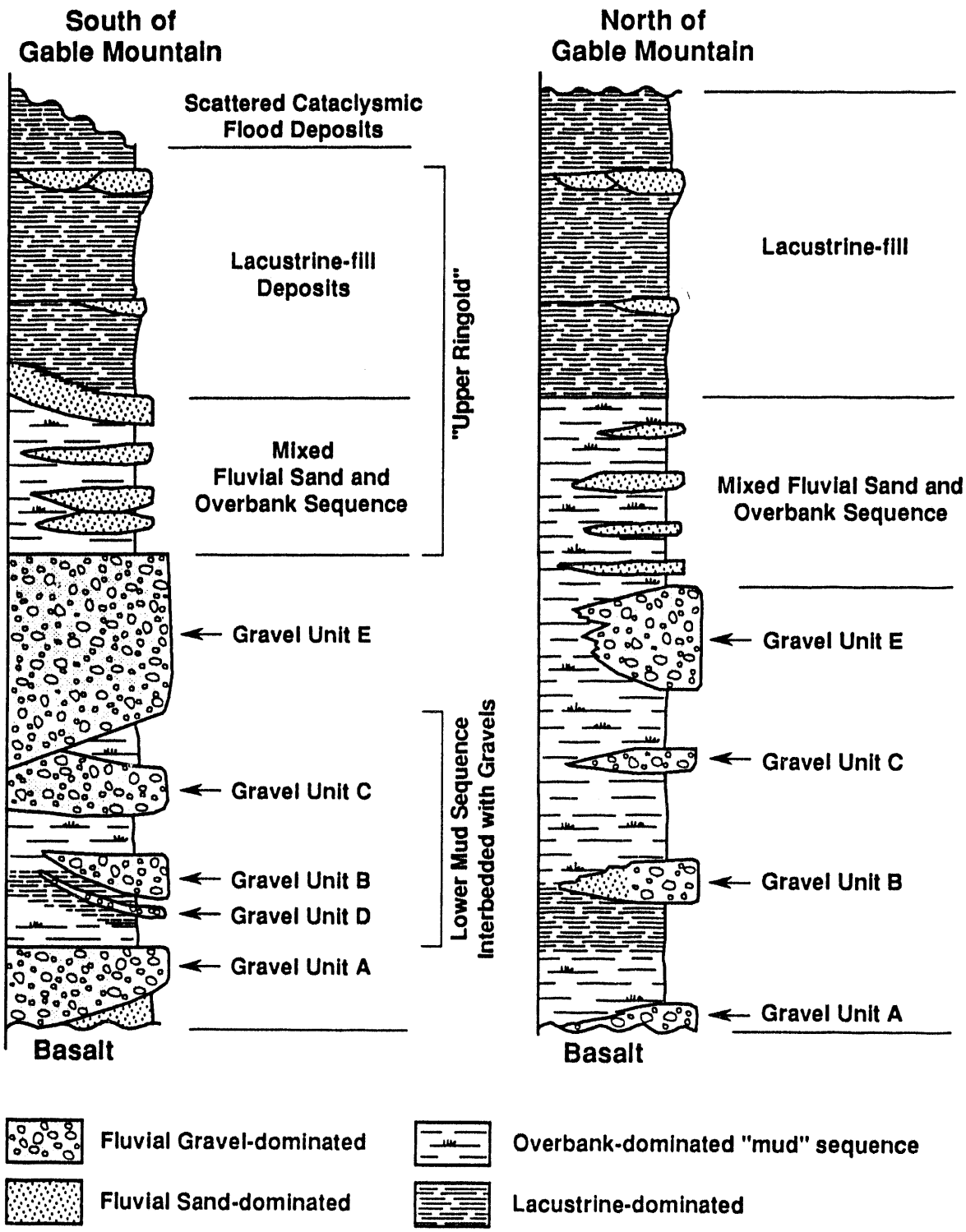


Figure 12. Generalized Stratigraphy of the Miocene-Pliocene Ringold Formation in the Pasco Basin. Figure Also Illustrates the Distribution of Major Sediment Facies in the Formation (Lindsey 1991a).



Several localized, informal units separate the Ringold Formation from the Hanford formation. These units are the (1) Plio-Pleistocene unit, (2) pre-Missoula gravels, and (3) early "Palouse" soil (see Figure 12) (Myers et al. 1979; Tallman et al. 1979, 1981; DOE 1988; Reidel et al. 1992). The Plio-Pleistocene unit and early "Palouse" soil consist of loess, pedogenic $CaCo_3$, and basaltic sands and gravels. Uncemented mixed lithology gravels with a quartzo-feldspathic matrix dominate the pre-Missoula gravels.

4.1.1.3.2 Hanford Formation. The Hanford formation consists of uncemented gravel, sand, and silt deposited by Pleistocene cataclysmic flood waters (Fecht et al. 1987; DOE 1988; Baker et al. 1989). The Hanford formation is thickest in the vicinity of the 200 West and 200 East Areas where it can be up to 107 m (350 ft) thick. The Hanford formation is divided into three facies (gravel, sand, and silt dominated) that are gradational with each other. The facies are summarized as follows:

(1) Gravel-dominated facies--Generally consists of cross-stratified, coarse-grained sand and granule to boulder gravel that contain minor intercalated silt-rich horizons. These gravels generally are uncemented and matrix poor, displaying an open-framework texture.

(2) Sand-dominated facies--Well-stratified, fine- to coarse-grained sand and granule gravel dominate. Silt content is variable, but where it is low an open-framework texture is common. Small pebbles and ripup clasts in addition to lenticular, pebble-gravel interbeds and silty interbeds may be present.

(3) Silt-dominated facies--Interbedded silt and fine- to coarse-grained sand forming well-stratified normally graded rhythmites are characteristic.

In addition to the three facies, clastic dikes also are commonly found in the Hanford formation as well as locally in other sedimentary units in the Pasco Basin (Black 1979). These clastic dikes are structures that generally cross-cut bedding, although they do locally parallel bedding. The dikes usually consist of thin, alternating vertical to subvertical layers of silt, sand, and granules. Where the dikes intersect the ground surface, a feature known as "patterned ground" can be observed.

4.1.1.4 Holocene Surficial Deposits. Holocene surficial deposits consist of silt, sand, and gravel that form a thin (4.9 m [<16 ft]) veneer across much of the Hanford Site. These sediments were deposited by a mix of eolian and alluvial processes.

4.1.1.5 Structural Geology. The Columbia Plateau is divided into three informal structural subprovinces: Blue Mountains, Palouse, and the Yakima Fold Belt (Reidel et al. 1989; Tolan and Reidel 1989). These structural subprovinces are delineated on the basis of their structural fabric. The Hanford Site is located in the eastern Yakima Fold Belt near its junction with the Palouse subprovince.

The Yakima Fold Belt consists of a series of segmented, narrow, asymmetric, and generally east-west trending anticlines that separate broad, low-amplitude structural basins (Reidel 1984, Reidel et al. 1989). The Pasco Basin (where the Hanford Site is situated) is one of the largest structural

basins within the Yakima Fold Belt. The Pasco Basin is bounded on the north by the Saddle Mountains anticline, on the west by the Hog Ranch-Naneum Ridge anticline, and on the south by the Rattlesnake Mountain anticline. The Palouse slope, a west-dipping monocline, bounds the Pasco Basin on the east. The Pasco Basin is divided into the Wahluke and Cold Creek synclines by the Gable Mountain anticline, the easternmost extension of the Umtanum Ridge anticline.

4.1.2 100-N Area Geology

The 100-N Area geology summarized in this section is drawn primarily from work done by Hartman (1993) on the RCRA Program for the 100-N Area. Most of the boreholes in the 100-N Area penetrate less than the upper 30 m (100 ft) of the stratigraphic section. Five boreholes near the 100-N Area penetrate the entire suprabasalt sediment sequence (Figure 13). One of these boreholes, 699-86-60, was drilled using a cable-tool rig; the driller's log is complete to the top of the CRBG. The other four deep boreholes were drilled by the Washington Public Power Supply System and are documented in WPPSS (1974). Three of the boreholes, 699-84-59 (called BH-16 in the WPPSS report), 699-81-62 (BH-17), and 699-86-64 (BH-18) were cored through the suprabasalt sequence and into the Asotin Member of the Saddle Mountains Basalt. The other deep borehole, 699-84-62A (BH-1), is a rotary-drilled hole to the top of basalt.

Figure 14 is a generalized stratigraphic column for the 100-N Area, which shows the units that are present in the area, approximate unit thicknesses, and the location of the water table.

Figure 15 shows the line of cross sections drawn west-east (Figure 16) and north-south (Figure 17) through the 100-N Area. These cross sections help to illustrate the stratigraphic location and areal distribution of units discussed in subsequent sections.

4.1.2.1 Columbia River Basalt Group. The cored boreholes that penetrate the CRBG near the 100-N Area encountered the following members of the Saddle Mountains Basalt: the Elephant Mountain Member, the Pomona Member, the Wilbur Creek Member, and the Asotin Member. The entire basalt stratigraphy of the area is interpreted from a borehole east of the site and another borehole north of the site in the Saddle Mountains. Columbia River basalts are estimated to be more than 3,658 m (12,000 ft) thick beneath the 100-N Area (Reidel and Hooper 1989).

The 100-N Area lies above a buried course of the ancestral Clearwater-Snake River, that existed during the Saddle Mountain time (Reidel and Fecht 1981). This channel developed along flow margins that controlled the river channel and the path of several basalt flows as well.

Figure 13. Locations of Washington Public Power Supply System Boreholes Near the 100-N Area (Hartman 1993).

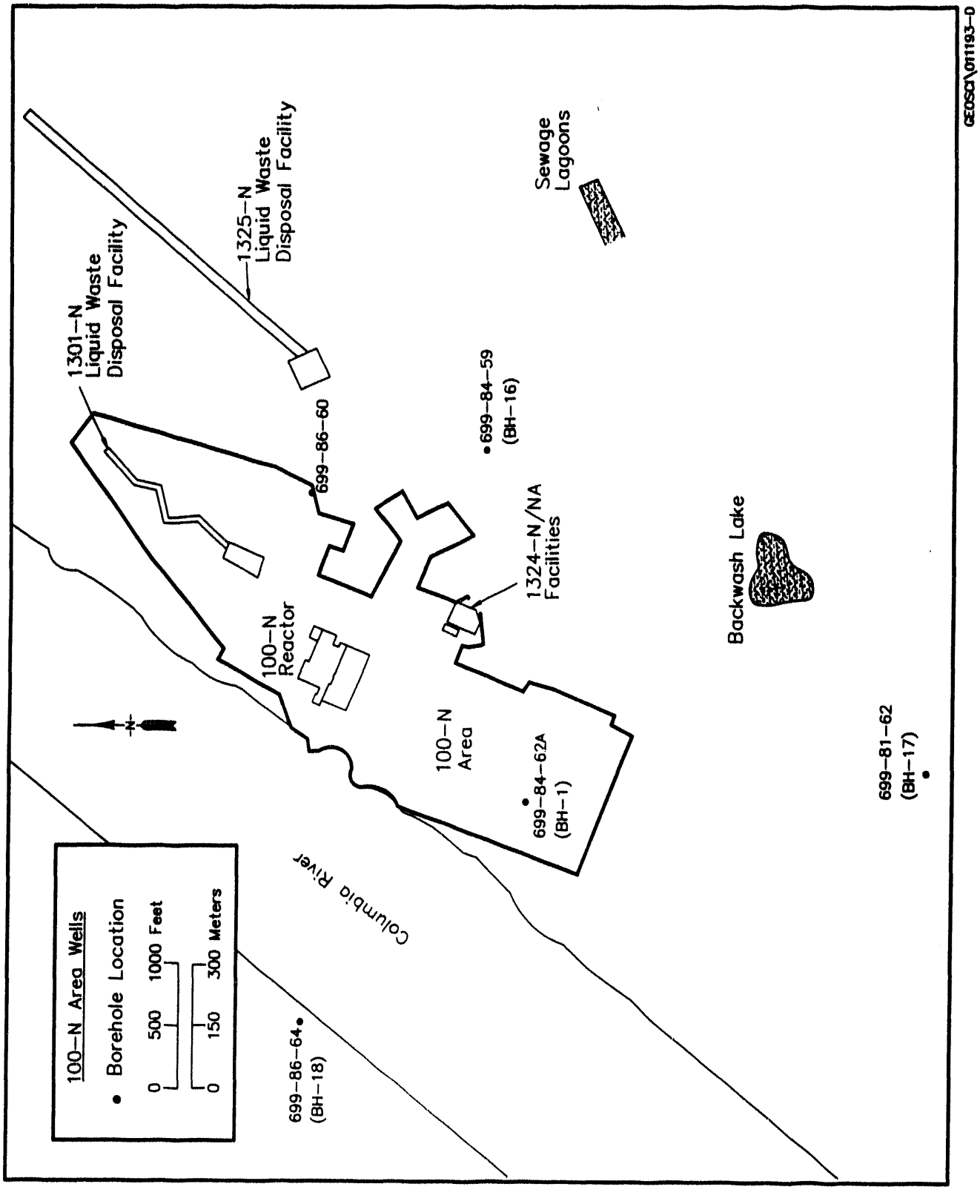


Figure 14. Generalized Stratigraphic Column for the 100-N Area.

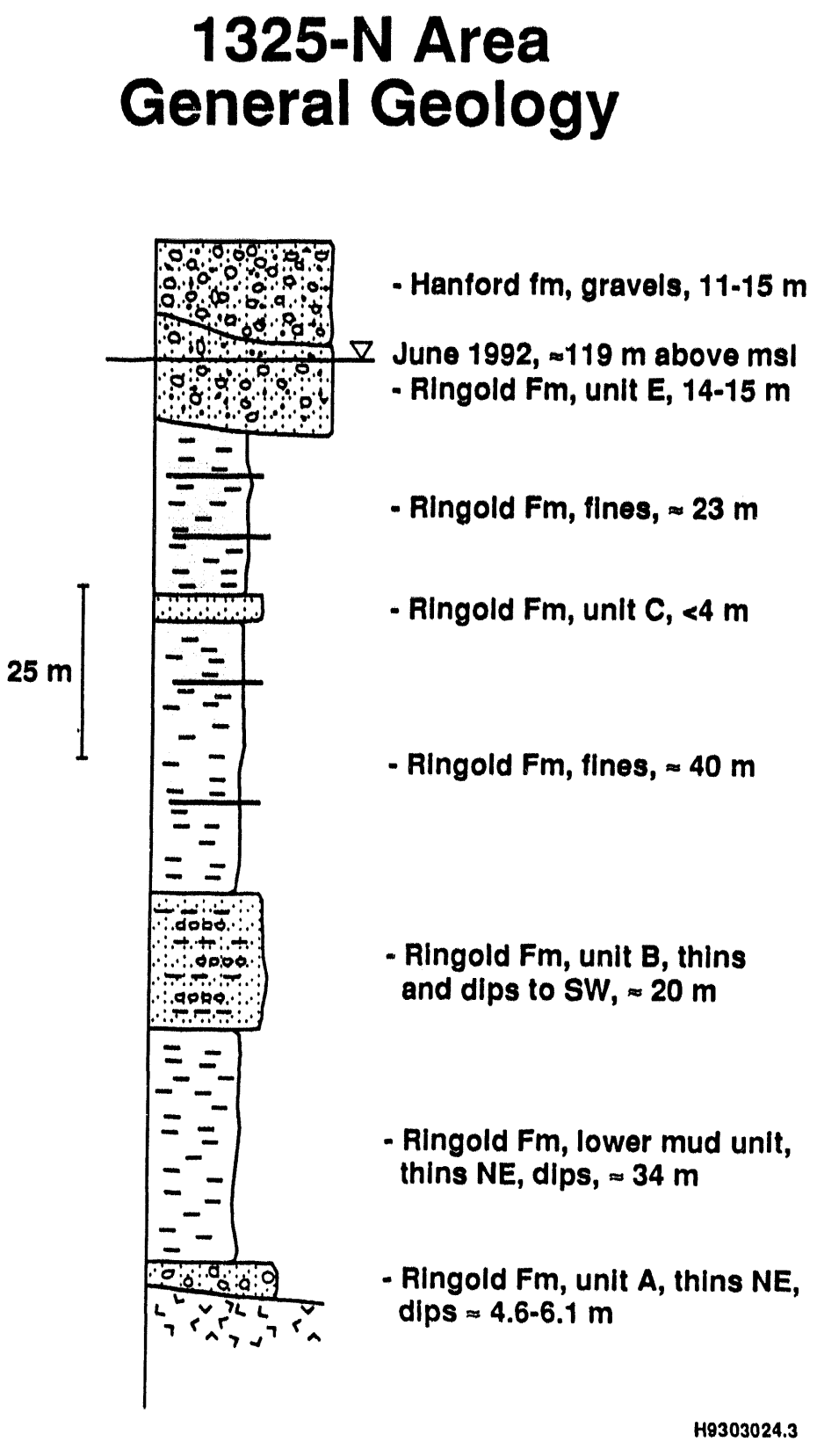
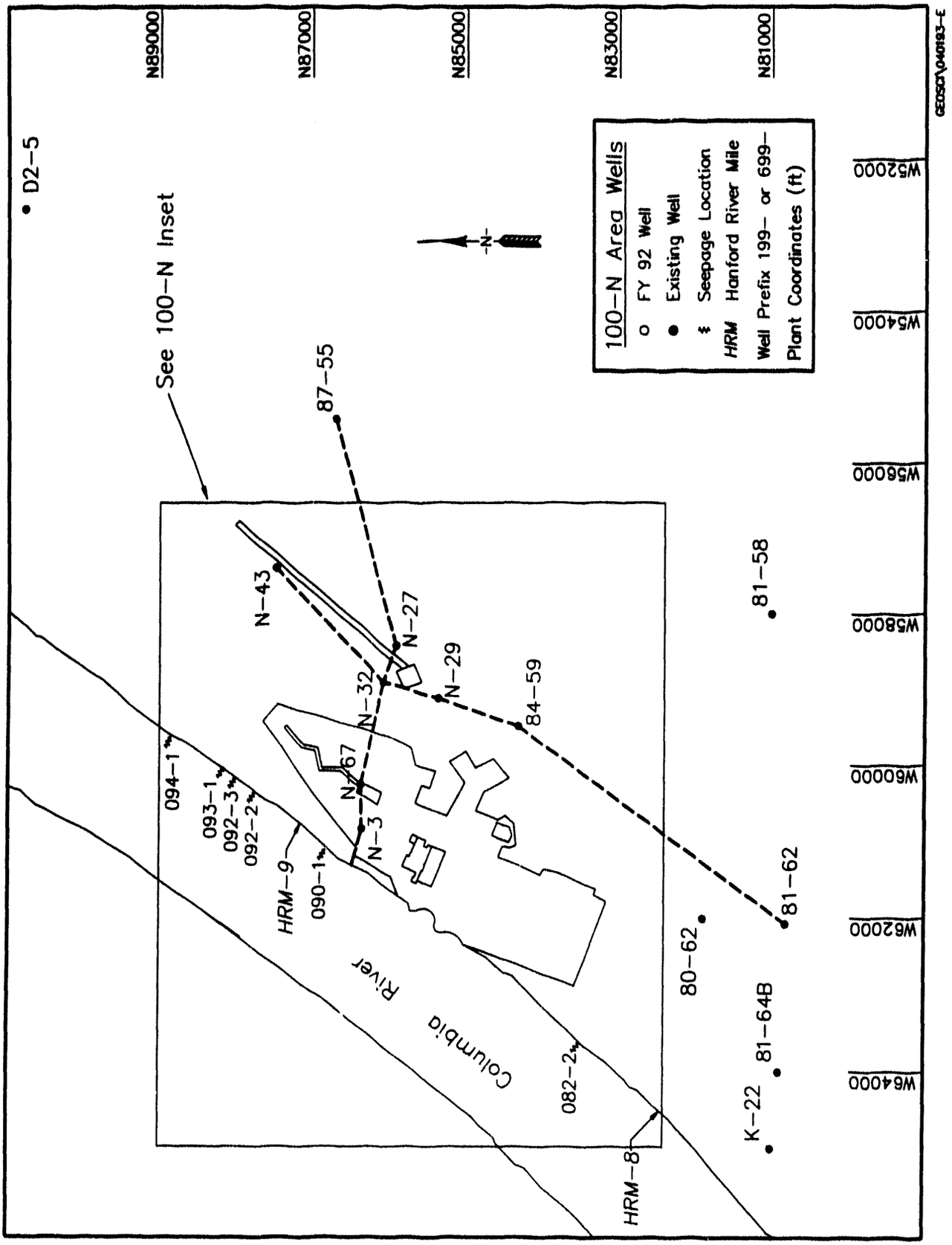


Figure 15. Line of Cross Sections Drawn West-East and North-South Through the 100-N and Surrounding Area.



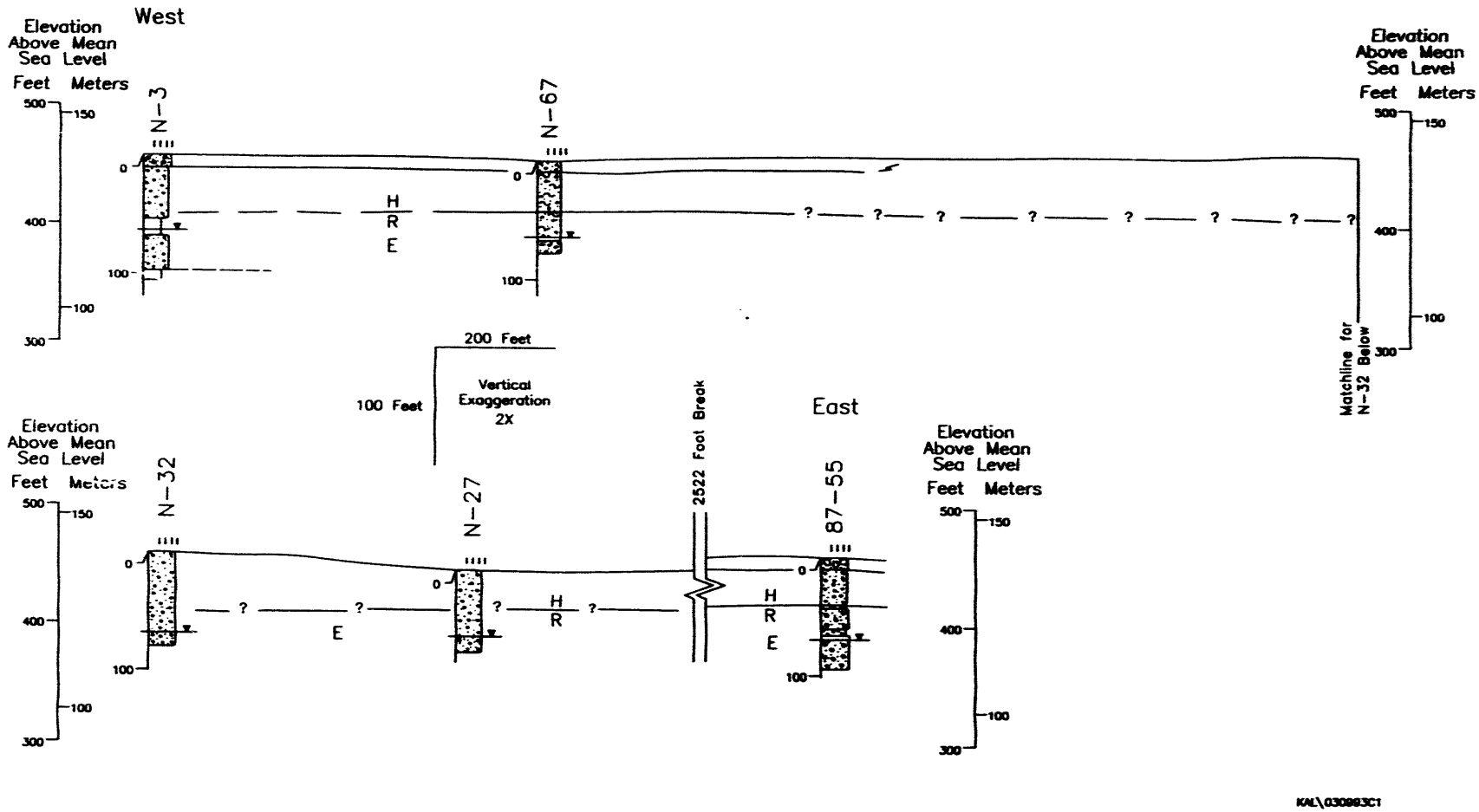


Figure 16. West-East Cross Section Through the 100-N and Surrounding Area.

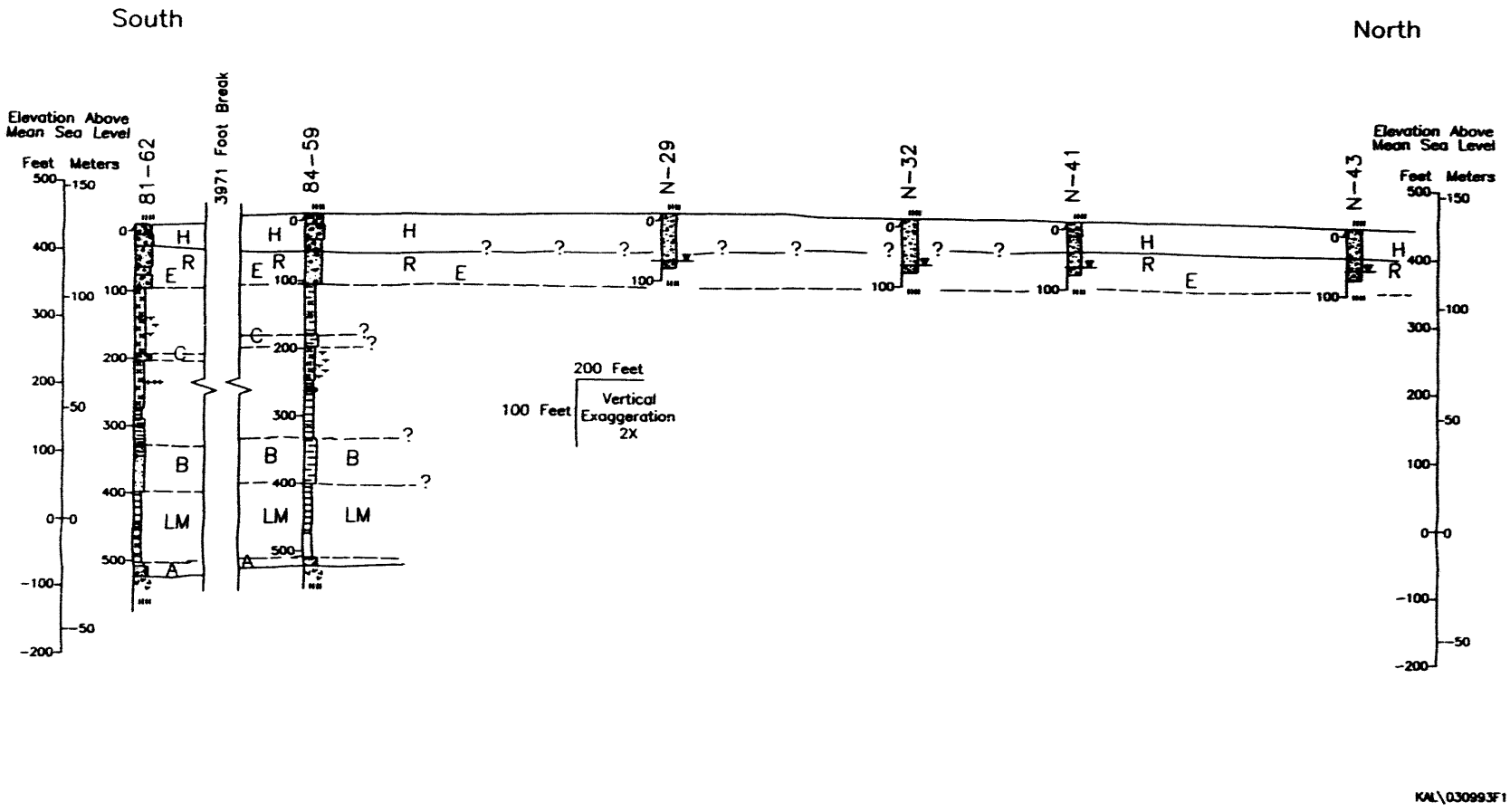


Figure 17. North-South Cross Section Through the 100-N and Surrounding Area.

4.1.2.2 Ellensburg Formation. Very little is known about the nature of the Ellensburg Formation in the 100-N Area. Three cored boreholes near the 100-N Area that penetrate the formation indicate the principal lithologies are sandstones, siltstones, and claystones, with minor conglomerates throughout the section. The lithologies probably represent channel and overbank deposits related to the ancestral Clearwater-Snake River system. The uppermost unit of the formation at the 100-N Area is the Rattlesnake Ridge interbed.

4.1.2.3 Suprabasalt Sediments.

4.1.2.3.1 Ringold Formation. The fluvial-lacustrine Ringold Formation was deposited in basins and generally east-west trending valleys by the ancestral Columbia River and its tributaries in response to development of the Yakima Fold Belt. Although exposures of the Ringold Formation are limited to the White Bluffs east of the 100-N Area and to the Smyrna and Taunton Benches north of the Saddle Mountains, data on the formation are available from the three cored boreholes and two rotary boreholes near the 100-N Area (see Figure 13).

The Ringold stratigraphy is best described in the 100-N Area in terms of facies type. At the 100-N Area, the formation consists predominantly of interstratified clays, silts, sands, and paleosols. Thin gravels occur near the top and bottom of the Ringold section.

A cross section through the 100 Areas (Figure 18), developed from the three cored and two rotary boreholes, shows the distribution of sediment types in the subsurface. The Ringold Formation is approximately 137 m (450 ft) thick at the 100-N Area, but thins to 129 m (422 ft) across the Columbia River to the west (at borehole 699-86-64). This may reflect erosion caused by the Columbia River, perhaps before the Pleistocene catastrophic flooding.

The contact between the Ringold Formation and the overlying Hanford formation in the 100-N Area is best distinguished using the compositional differences in the sands and gravels of the two units. The Ringold Formation is dominated by quartzo-feldspathic materials and the Hanford formation is dominated by basaltic material. In the field, this is best recognized as a color change; the Hanford formation is black or grey, and the Ringold Formation is tan or brown.

4.1.2.3.2 Hanford Formation. The Hanford formation represents proglacial flood deposits, primarily from the late-Pleistocene. Alluvial deposits include river deposits, landslides, and windblown loess and sand.

In the 100-N Area the Hanford formation consists of sands and gravels. There appears to be a general coarsening downward, but no detailed work has been done in the area on this topic. The unit is approximately 12 to 18 m (40 to 60 ft) thick. The elevation of the Hanford-Ringold contact, and where known the base of the uppermost aquifer, are given in Table 5.

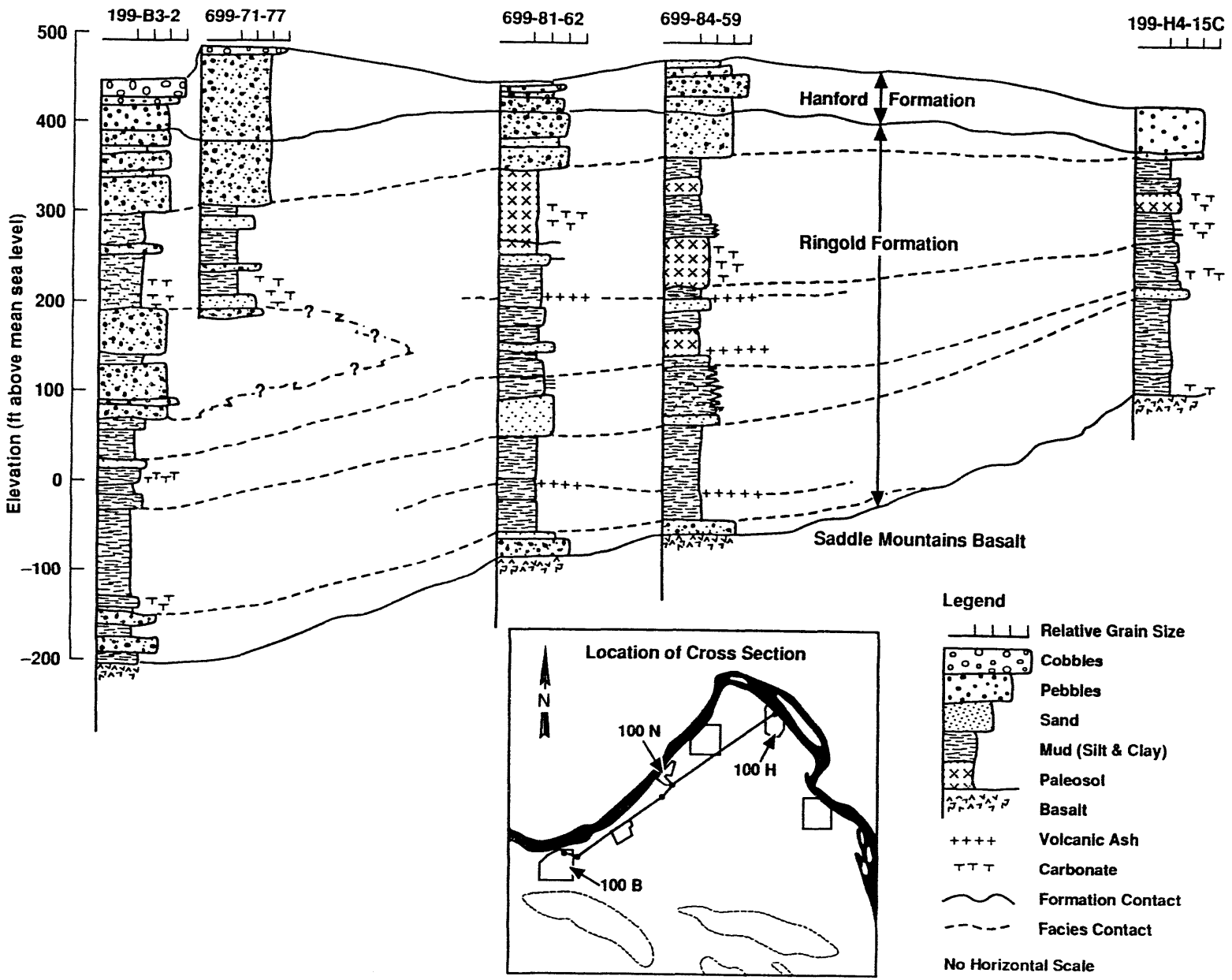


Figure 18. Geologic Cross Section through the 100 Areas.

Table 5. Elevation of Geologic Units at 100-N Area Resource Conservation and Recovery Act of 1976 Sites (Hartman 1993).

Site	Elevation of surface - m msl (ft msl)	Elevation of Hanford/Ringold contact - m msl (ft msl)	Elevation top of fine-grained unit - m msl and (ft msl) [no. of wells]
1301-N	137 to 140 (450 to 460)	120 to 127 (394 to 417)	107 to 110 (350 to 360 [4])
1325-N	137 to 140 (450 to 460)	124 to 126 (406 to 415)	107 (350 [1])

msl = mean sea level.

4.1.2.4 Holocene Surficial Deposits. Holocene deposits in the vicinity of the 100-N Area consist of Columbia River overbank sediments and minor amounts of eolian silt and sand.

4.1.3 Regional and Hanford Site Hydrology

4.1.3.1 Surface Water. Surface drainage enters the Pasco Basin from several other basins, which include the Yakima River Basin, Horse Heaven Basin, Walla Walla River Basin, Palouse/Snake Basin, and Big Bend Basin (Figure 19) (DOE 1988). Within the Pasco Basin, the Columbia River is joined by major tributaries including the Yakima, Snake, and Walla Walla Rivers. No perennial streams originate within the Pasco Basin. Columbia River inflow to the Pasco Basin is recorded at the United States Geological Survey gage below Priest Rapids Dam, and outflow is recorded below McNary Dam. Average annual flow at these recording stations is approximately 1.1×10^{11} m^3 (8.7×10^7 acre-ft) at the United States Geological Survey gage and 1.6×10^{11} m^3 (1.3×10^8 acre-ft) at the McNary Dam gage (DOE 1988).

Total estimated precipitation over the basin averages less than 15.8 cm/yr (6.2 in./yr). Mean annual runoff from the basin is estimated to be less than 3.1×10^7 m^3/yr (1.1×10^9 ft^3/yr), or approximately 3% of the total precipitation. The remaining precipitation is assumed to be lost through evapotranspiration with a small component (perhaps less than 1%) recharging the groundwater system (DOE 1988).

Primary surface water features associated with the Hanford Site are the Columbia River and the Yakima River. West Lake, about 40,470 m^2 (10 acres) in size and less than 1 m (3 ft) deep, is the only natural lake within the Hanford Site (DOE 1988). Wastewater ponds, cribs, and ditches associated with nuclear fuel processing and waste disposal activities also are present on the Hanford Site (Figure 20).

Figure 19. Hydrologic Basins Designated for the Washington State Portion of the Columbia Plateau (DOE 1988).

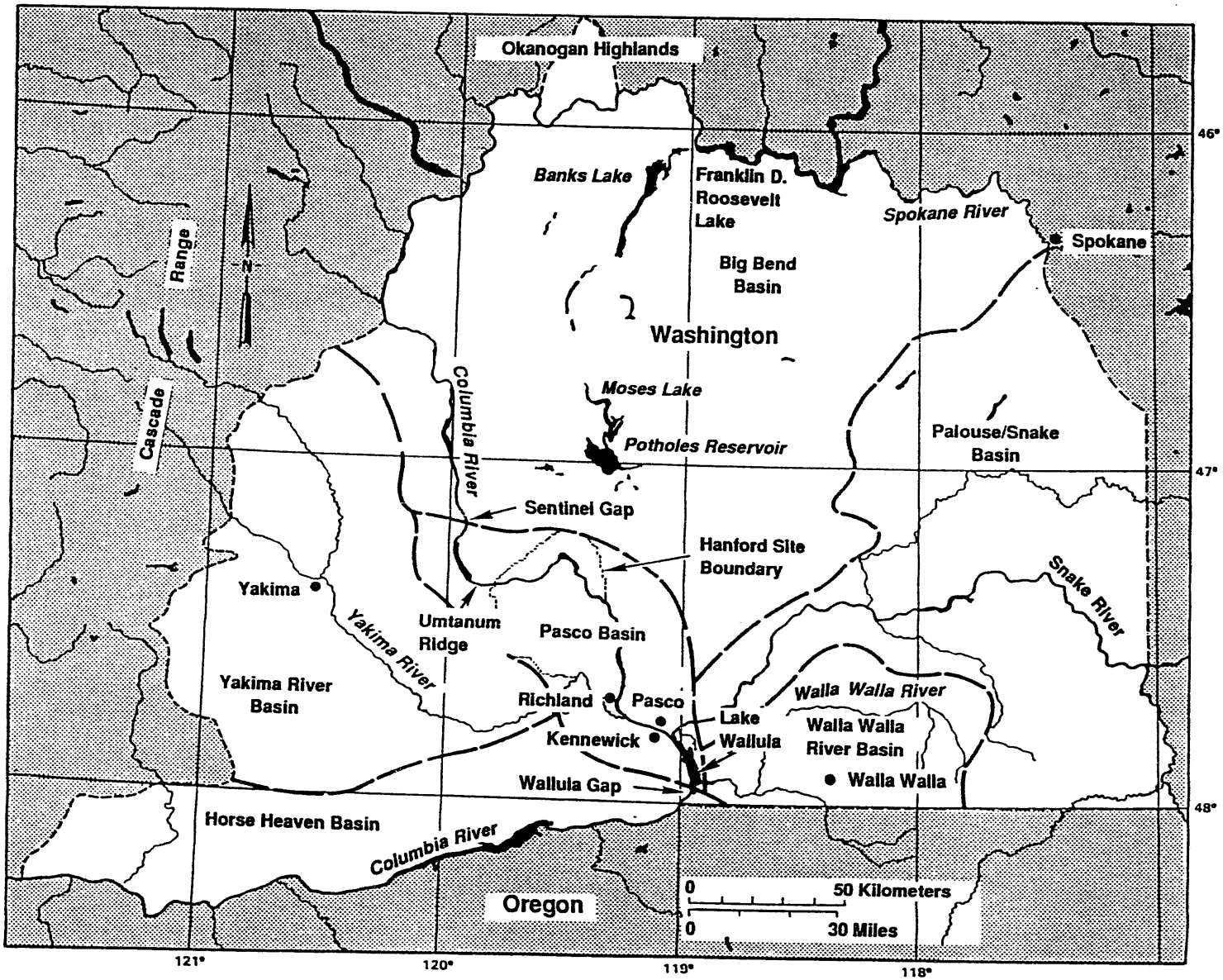


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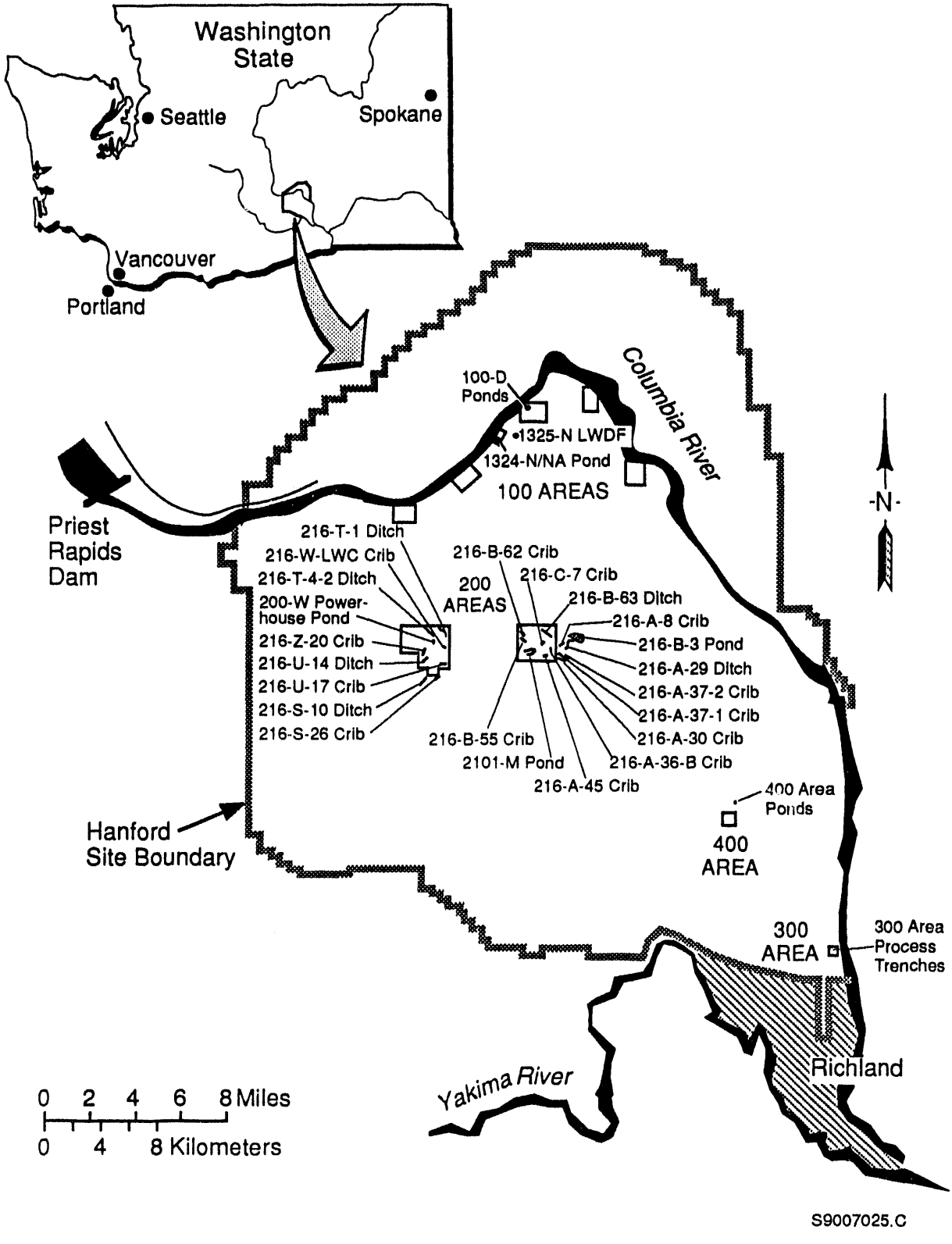
Figure 20. Location of Major Liquid Effluent Disposal Sites on the Hanford Site.



The Columbia River flows through the northern part and along the eastern border of the Hanford Site. This section of river, the Hanford Reach, extends from Priest Rapids Dam to the headwaters of Lake Wallula (the reservoir behind McNary Dam). Flow along the Hanford Reach is controlled by the Priest Rapids Dam. Several drains and intakes also are present along the Hanford Reach, including irrigation outfalls from the Columbia Basin Irrigation Project, the Washington Public Power Supply System Nuclear Project 2 intakes, and Hanford Site intakes for onsite water use.

Routine water quality monitoring of the Columbia River is conducted by the DOE for both radiological and nonradiological parameters and has been reported by the Pacific Northwest Laboratory since 1973. Ecology has issued a Class A (excellent) quality designation for Columbia River water along the Hanford Reach from Grand Coulee Dam, through the Pasco Basin, to McNary Dam. This designation requires that all industrial uses of this water be compatible with other uses, including drinking, wildlife habitat, and recreation. In general, the Columbia River water is characterized by a very low suspended load, a low nutrient content, and an absence of microbial contaminants (DOE 1988).

Approximately one-third of the Hanford Site is drained by the Yakima River system. Cold Creek and its tributary, Dry Creek, are ephemeral streams within the Yakima River drainage system. Both streams drain areas along the western part of the Hanford Site and cross the southwestern part of the Hanford Site toward the Yakima River. Surface flow, which may occur during spring runoff or after heavier-than-normal precipitation, infiltrates and disappears into the surface sediments. Rattlesnake Springs, located on the western part of the Hanford Site, forms a small surface stream that flows for about 2.9 km (1.8 mi) before infiltrating into the ground.

4.1.3.2 Groundwater. Information in this section is summarized from Delaney et al. (1991). The hydrogeology of the Pasco Basin is characterized by a multiaquifer system that consists of four hydrogeologic units that correspond to the upper three formations of the CRBG (Grande Ronde Basalt, Wanapum Basalt, and Saddle Mountains Basalt) and the suprabasalt sediments. The basalt aquifers consist of the tholeiitic flood basalts of the CRBG and relatively minor amounts of intercalated fluvial and volcaniclastic sediments of the Ellensburg Formation. Confined zones in the basalt aquifers are present in the sedimentary interbeds and/or interflow zones that occur between dense basalt flows. The main water-bearing portions of the interflow zones are networks of interconnecting vesicles and fractures of the flow tops and bottoms (DOE 1988). The suprabasalt sediment or uppermost aquifer system consists of fluvial, lacustrine, and glaciofluvial sediments. Regionally, this aquifer is unconfined and is contained largely within the Ringold Formation and Hanford formation. Table 6 presents hydraulic parameters for various water-bearing geologic units at the Hanford Site.

Local recharge to the shallow basalt aquifers results from infiltration of precipitation and runoff along the margins of the Pasco Basin and in areas of artificial recharge where a downward gradient from the unconfined aquifer system to the uppermost confined basalt aquifer may occur. Regional recharge of the deep basalt aquifers is inferred 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).

Table 6. Hydraulic Parameters for Various Areas at the Hanford Site (Delaney et al. 1991).

Location	Interval tested	Hydraulic conductivity m/d (ft/d)	Transmissivity m^2/d (ft^2/d)	Effective porosity	Data source
100 Areas	Rattlesnake Ridge interbed	(0 to 100)	--	<10%	Gephart et al. (1979)
Hanford Site	Saddle Mountain Basalt Flowtop	(10^{-2} to 10^{-6})	--	5%	Cushing (1989)
100 Areas	Ringold Formation Unit E	(29 to 1,297)	(5,750 to 26,700)	--	Liikala et al. (1988)
200 Areas	Rattlesnake Ridge interbed	--	(8 to 1,165)	--	Graham et al. (1984)
200 East Area	Elephant Mountain Interflow Zone	--	(7.5 to 6,120)		Graham et al. (1984)
Hanford Site	Selah interbed	--	(3×10^{-5})	--	Graham et al. (1984)
200 West Area	Ringold Formation Unit E	(0.6 to 200)	--	--	Last et al. (1989)
1100 Area	Ringold Formation Units C/B	(3×10^{-1} to 5)	--	--	Lindberg and Bond (1979)
1100 Area	Ringold formation overbank Deposits	(8×10^{-4} to 1×10^{-1}	--	--	Lindberg and Bond (1979)
300 Area	Levey interbed	(0.01 to 1,000)	--	--	DOE-RL (1990)
300 Area	Ringold Formation	(1.9 to 10,000)	--	--	DOE-RL (1990)
300 Area	Hanford formation	(11,000 to 50,000)	--	--	DOE-RL (1990)

Groundwater discharge from shallow basalt aquifers is probably to the overlying aquifers and to the Columbia River. The discharge area(s) for the deeper groundwater system is (are) uncertain, but flow is inferred to be generally southeastward, with discharge to the south of the Hanford Site (DOE 1988).

Erosional "windows" through dense basalt flow interiors allow direct interconnection between the uppermost aquifer system and underlying confined basalt aquifers. Graham et al. (1984) reported that some contamination was present in the uppermost confined aquifer (Rattlesnake Ridge interbed) south and east of Gable Mountain Pond. Graham et al. (1984) evaluated the hydrologic relationships between the Rattlesnake Ridge interbed aquifer and the unconfined aquifer in this area, and delineated a potential area of intercommunication beneath the northeast portion of the 200 East Area.

The uppermost aquifer system is regionally unconfined beneath the Hanford Site and lies at depths ranging from less than 0.3 m (1 ft) below ground surface near West Lake and the Columbia and Yakima Rivers, to greater than 107 m (350 ft) in the central portion of the Cold Creek syncline. Groundwater in this aquifer system occurs within the glaciofluvial sands and gravels of the Hanford formation and the fluvial/lacustrine sediments of the Ringold Formation. Ringold sediments are divided into five lithofacies: (1) fluvial gravel, (2) fluvial sand, (3) overbank deposits consisting of silt and sand, (4) lacustrine deposits, and (5) basaltic debris flow gravel (see Figure 12).

The position of the water table in the southwestern Pasco Basin is generally within Ringold fluvial gravels of unit E. In the northern and eastern Pasco Basin the water table generally is within the Hanford formation. Hydraulic conductivities for the Hanford formation (610 to 3,048 m/day [2,000 to 10,000 ft/day]) are much greater than those of the gravel facies of the Ringold Formation (186 to 930 m/day [610 to 3,050 ft/day]) (Graham et al. 1981). The main body of the unconfined aquifer generally occurs within the Ringold Formation.

The base of the uppermost aquifer system is defined as the top of the uppermost basalt flow. However, fine-grained overbank and lacustrine deposits in the Ringold Formation locally form confining layers for Ringold fluvial gravels (units A, B, D, C) underlying unit E. The uppermost aquifer system is bounded laterally by anticlinal basalt ridges and is approximately 152 m (500 ft) thick near the center of the Pasco Basin.

Sources of natural recharge to the uppermost aquifer system are rainfall and runoff from the higher bordering elevations, water infiltrating from small ephemeral streams, and 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 (Gee 1987; Routson and Johnson 1990; Rockhold et al. 1990). Conclusions from these studies vary. Gee (1987) and Routson and Johnson (1990) concluded that little, if any, downward percolation of precipitation occurs on the 200 Area Plateau where the sediments are layered and vary in texture, and that all moisture penetrating the soil is removed by evapotranspiration. Rockhold et al. (1990) suggested that downward water movement below the root zone is common in the 300 Area, where soils are coarse textured.

Artificial recharge of the uppermost aquifer system occurs from the disposal of large volumes of wastewater on the Hanford Site (principally in the 200 Areas), and from large irrigation projects surrounding the Hanford Site. Figures 21 and 22 illustrate the groundwater table for the Hanford Site during the periods of January 1944 and June 1989. Effluent disposal at the Hanford Site altered these hydraulic gradients and flow directions. Before operations at the Hanford Site began in 1944, the hydraulic gradient in all but the southwesternmost portion of the Hanford Site was approximately 1.5 m/km (5 ft/mi). Regional groundwater flow generally was toward the east-northeast, although flow north of Gable Mountain was more to the north. Groundwater flow north of Gable Mountain now trends in a more northeasterly direction resulting from mounding near reactors and flow through Gable Gap. Flow south of Gable Mountain is redirected locally by the groundwater mounds in the 200 Areas. There is also a component of groundwater flow to the north between Gable Mountain and Gable Butte from the 200 Areas.

Wastewaters discharged on the Hanford Site have contaminated large areas of the unconfined aquifer and a portion of the confined aquifer (Rattlesnake Ridge interbed). The primary contaminants that have reached the upper confined aquifer and the uppermost aquifer system are tritium, iodine-129, ruthenium-106, technetium-99, uranium, nitrate, and chromium (DOE 1987). The groundwater is routinely and extensively monitored to record the movement of contaminants and to determine any impact from the Hanford Site to the public. Groundwater monitoring reports are produced annually (e.g., Serkowski and Jordan 1989).

Temporary reversal of groundwater flow entering the Columbia River may occur during transient, high-river stages. This occurrence is known as bank storage. Newcomb and Brown (1961) made correlations between groundwater level and river-stage fluctuations along a 81-km (50-mi) reach of the Columbia River adjacent to the Hanford Site. Newcomb and Brown (1961) concluded that a 260-km^2 (100-mi^2) area within the Hanford Site was affected by bank storage. During a 45-day rise in river stage, it was estimated that water infiltrated at an average rate of 111 m^3/day (3,700 acre-ft/day) versus 30 m^3/day (1,000 acre-ft/day) during the 165-day recession period (time between consecutive storage events). In subsequent years dam control on the Columbia River has reduced the magnitude of bank storage fluctuations on the groundwater system.

4.1.4 100-N Area Hydrology

The hydrology summarized in this entire section was taken from the work of Hartman (1993) on the 100-N RCRA sites.

Figure 21. Hindcast Water Table Map of the Hanford Site, January 1944 (ERDA 1975).

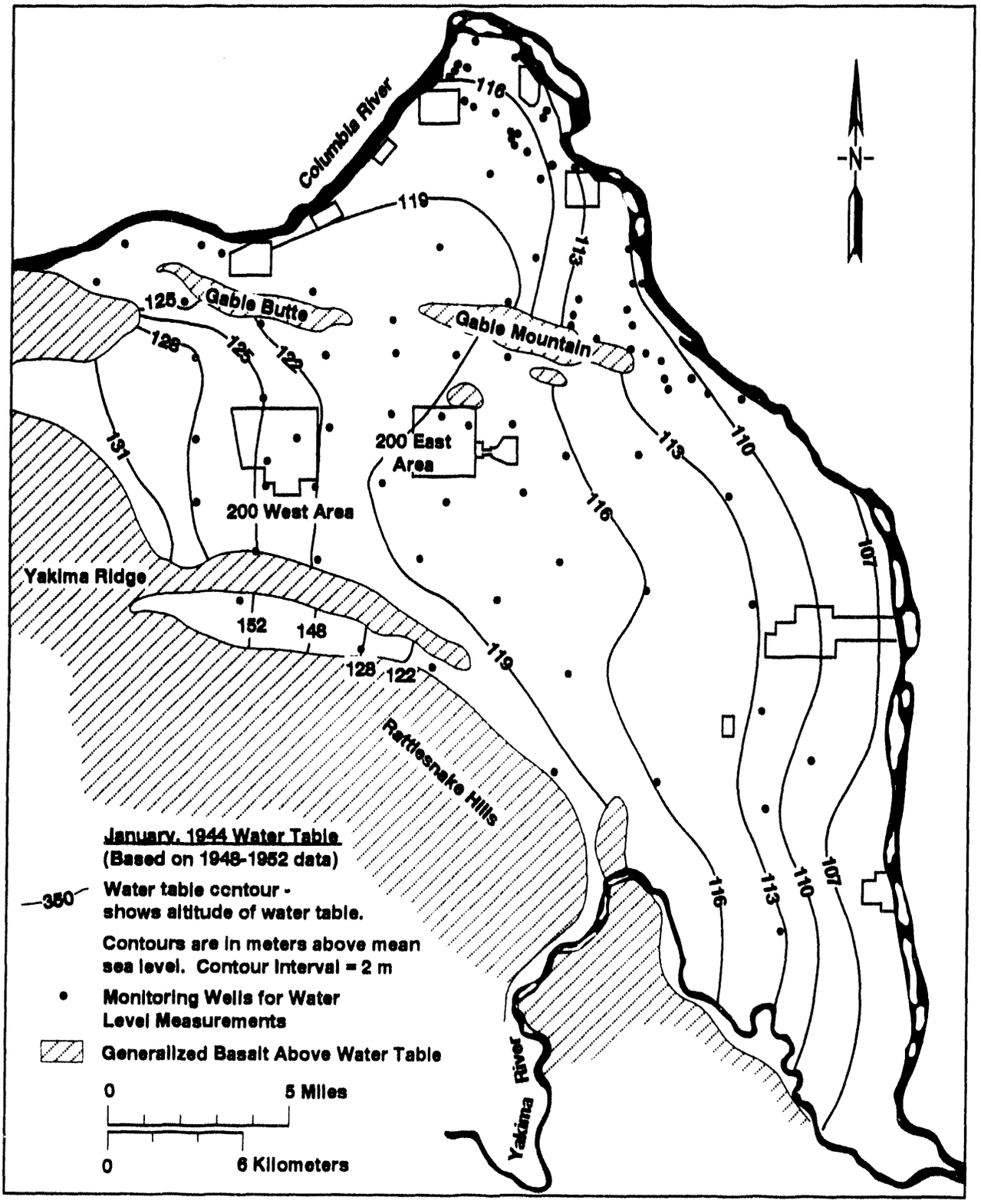


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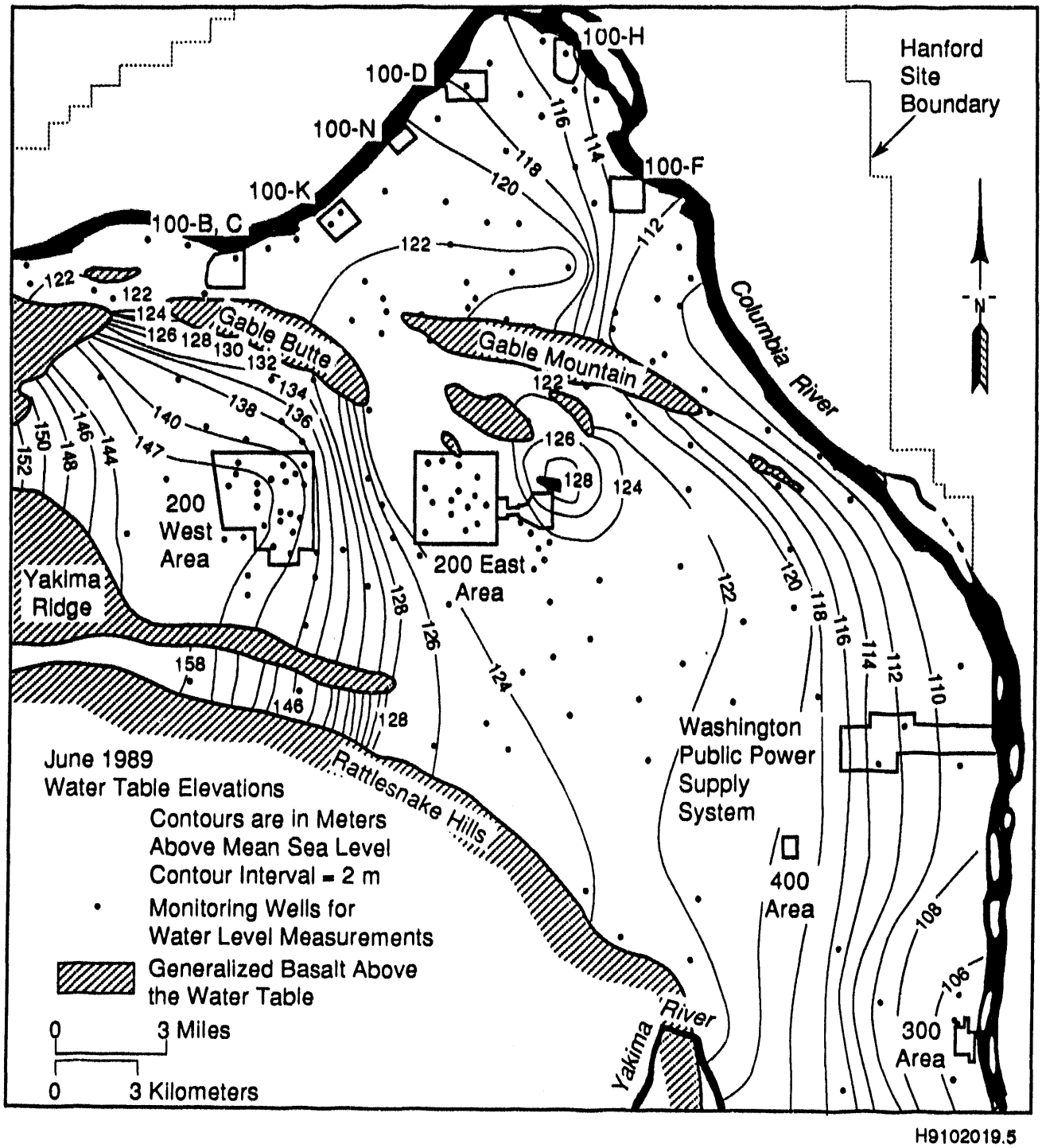
Figure 22. Hanford Site Water Table Map, June 1989 (Smith et al. 1989).



H9102019.5

Most of the wells in the 100-N Area are relatively shallow, so information regarding hydrologic conditions at greater depths is limited. The current conceptualization of the 100-N Area hydrostratigraphy is shown in Figure 23. It comprises the following major hydrostratigraphic units:

- Surface water
- Vadose zone
- Unconfined aquifer
- Ringold confined aquifer system
- Basalt and interbed aquifer system.

In the following discussion and in the rest of this document, 100-N Area well numbers, normally written as 199-N-#, will be abbreviated N-#. Monitoring well locations are shown in Figure 24.

4.1.4.1 Surface Water. The Columbia River is the only natural surface water feature affecting the 100-N Area. The river stage is regulated by the dams upstream, and daily fluctuations of 2 m (6 ft) are common. The river stage usually fluctuates 2 to 3 m (8 to 10 ft) through the year. Continuous river stage measurements are available from the 100-N Gaging Station.

The following waste treatment and liquid effluent disposal units in the 100-N Area have been sources of artificial groundwater recharge (see Figure 13). As a result, mounds in the water table developed beneath these disposal sites and local groundwater flow patterns were altered.

- 1301-N LWDF--An unlined crib and trench that was used for disposal of cooling water from the N Reactor from 1963 to 1985. Discharge rate was approximately 6,435 L/min (1,700 gal/min).

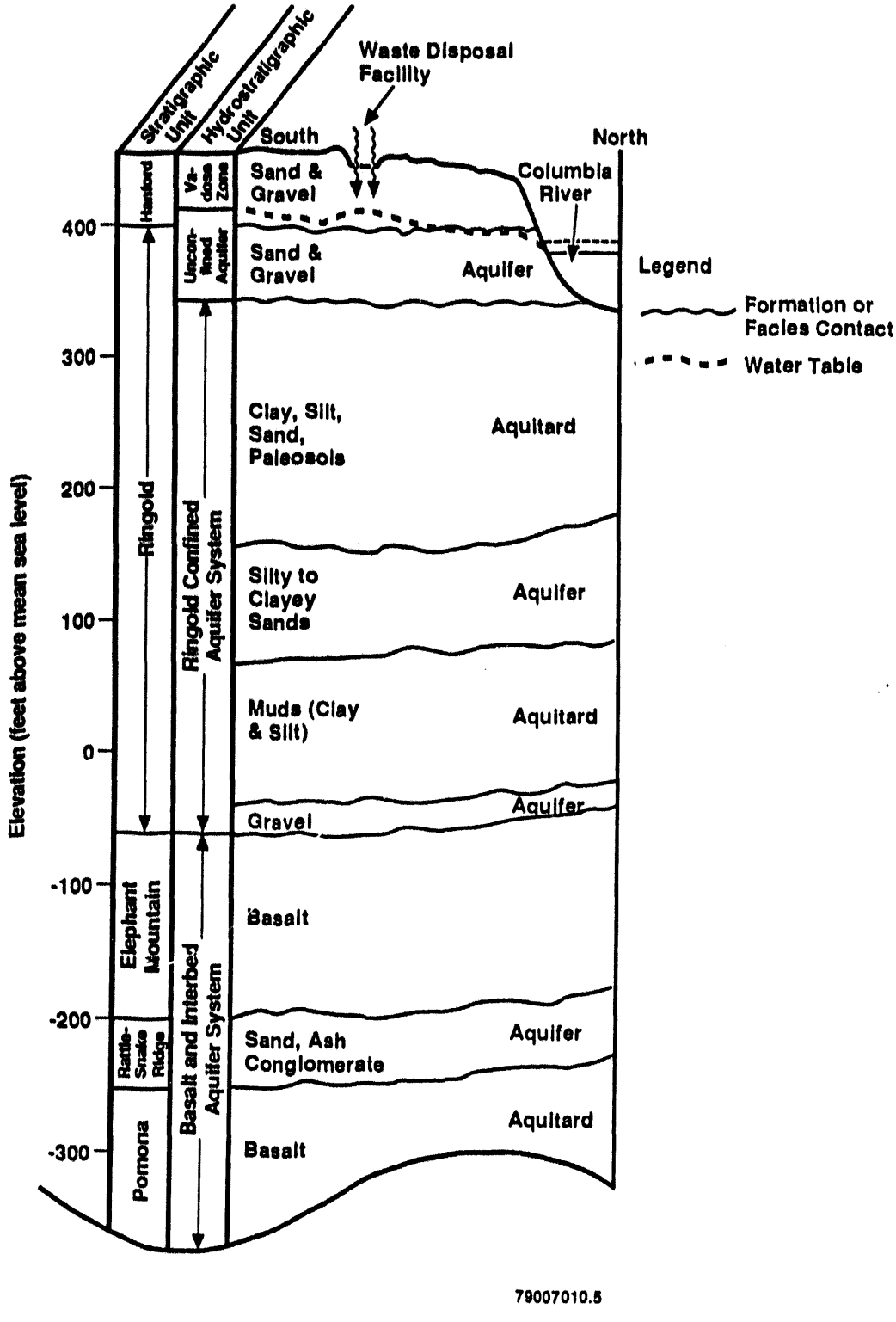
- 1325-N LWDF--Replaced 1301-N LWDF in 1985. Total flow to the 1325-N LWDF during normal reactor operation was 3,407 to 6,057 L/min (900 to 1,600 gal/min). Current discharge rate is zero (since April 1991).

- 1324-NA Percolation Pond--An unlined pond that received wastes from 1977 to August 1990. Discharge rates averaged approximately 1,136 L/min (300 gal/min) when the pond was receiving filter backwash water as well as regeneration wastes and cooling water. The filter backwash was rerouted to another facility in 1983.

- Backwash Lake--Disposal site for nonregulated waste from the 183-N Filtered Water Plant. Has received effluent since 1983. Estimated volume of discharge averaged 1,135,620 L/day (300,000 gal/day). Current and expected future discharge rate averages approximately 42 L/min (11 gal/min).

- 100-N Area Sewage Lagoon--Three ponds (aeration, stabilization, and infiltration) that are used to treat the waste discharged to the 100-N Sanitary Sewer System (Figure 25). Waste is also trucked in from septic tanks in the Hanford Site. Both trucked-in and piped-in wastes are discharged directly into the aeration pond (WHC 1993).

Figure 23. Conceptual Drawing of Hydrogeologic Units in the 100-N Area (Hartman 1993).



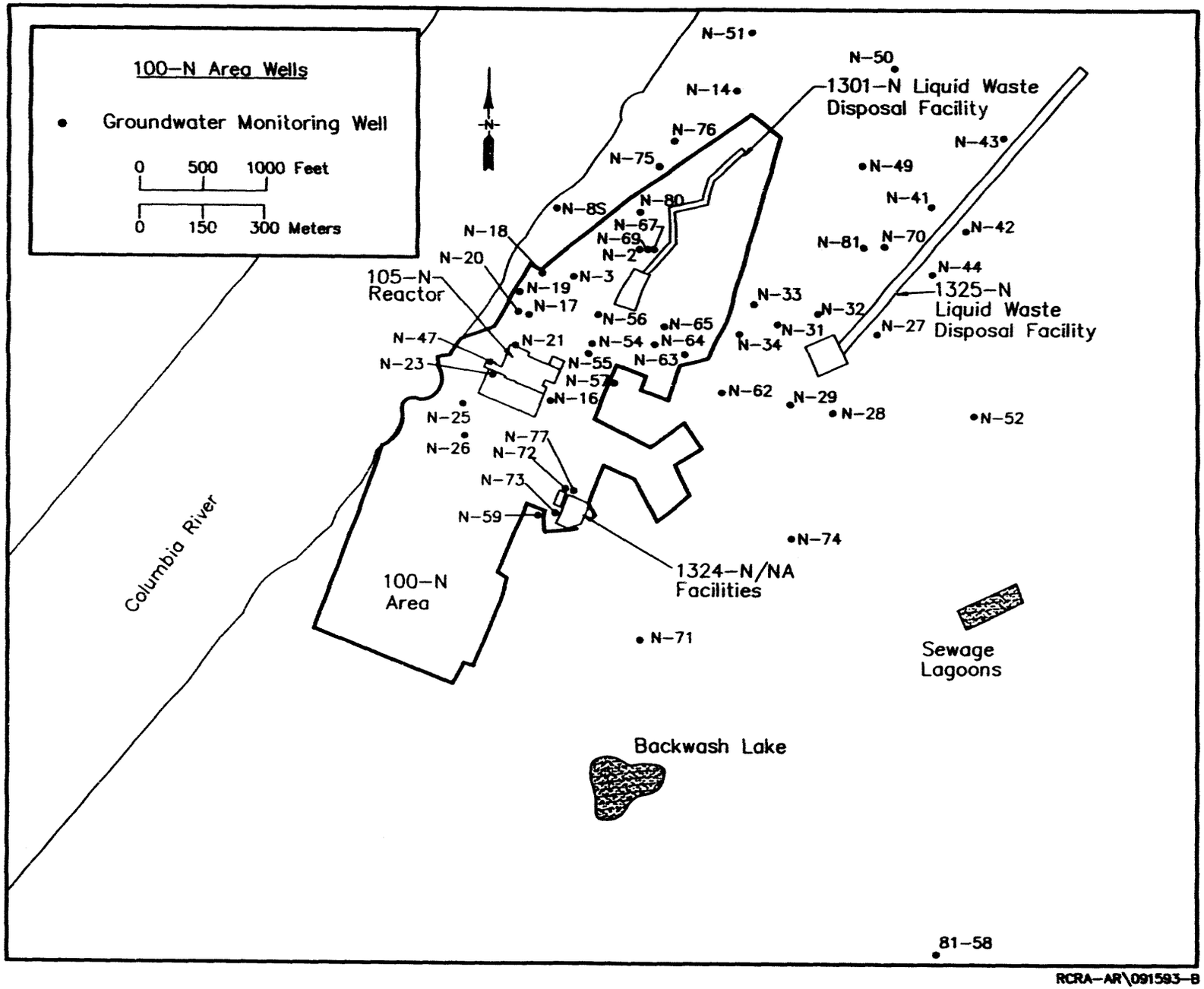


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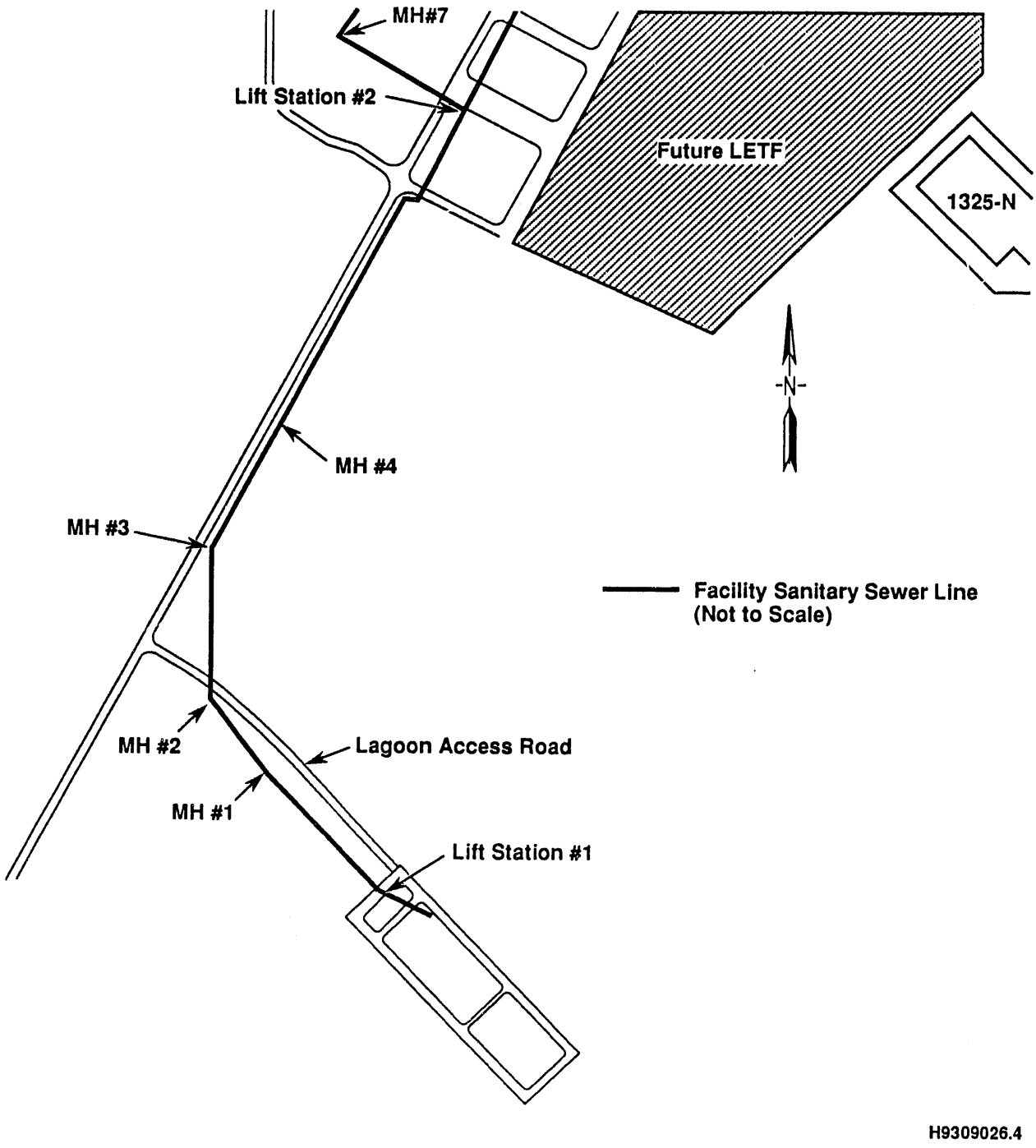
Figure 24. Groundwater Monitoring Wells In and Near the 1325-N Liquid Waste Disposal Facility.

Figure 25. 100-N Area Sewage Lagoon (WHC 1993).



Locally, groundwater flows toward the Columbia River. However, during times of high river stage, water may flow from the river into the aquifer. This reversed gradient occasionally extends as far inland as the 1301-N LWDF (see Figure 13).

Groundwater also enters the Columbia River in the 100-N Area via a series of seeps, known collectively as "N Springs". They are located along the bank of the river, adjacent to the 1301-N LWDF, and extend approximately 2.7 km (1.7 mi) downstream. These seeps, particularly those located nearest the 100-N Area, appear to be related to increased groundwater levels resulting from effluent discharged to the ground. Now that effluent discharge to the ground virtually has ceased, there are fewer seeps. The remaining seeps discharge at a lower rate. When the river stage drops, return flow from bank storage also contributes to the N Springs discharges. Water samples from N Springs are analyzed annually. Recent spring sampling occurred as part of the 100 Areas *Comprehensive Environmental Response, Compensation, and Liability Act of 1980* (CERCLA) remedial investigations. The sampling focused on chemical contaminants as well as radionuclides (DOE-RL 1992c). It produced new data for many chemical indicators of Hanford Site contaminants, more widespread observations of chemical and radionuclide distributions, and an additional data set from which trend information can be derived. Locations of the seepage areas sampled are shown in Figure 26 (Peterson and Johnson 1992).

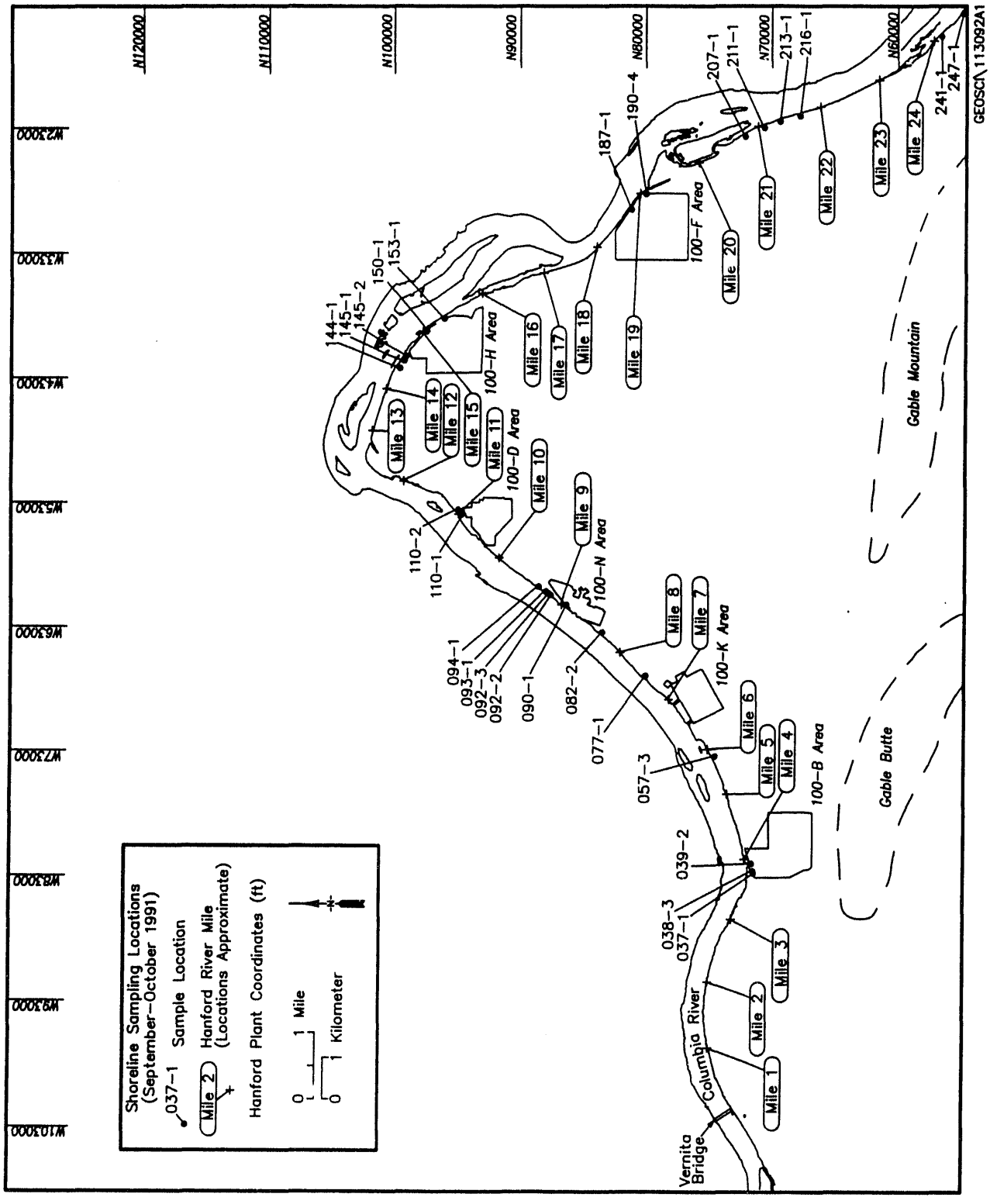
4.1.4.2 Vadose Zone. The vadose zone beneath the 100-N Area comprises primarily unconsolidated sediments of the Hanford formation. This highly permeable unit is composed mainly of cobbles, boulders, gravel, and coarse sand. Drilling data indicate that isolated lenses of silty sand and gravel are also present. The vadose zone also includes the top few feet (couple of meters) of the Ringold Formation in some parts of the area. These sediments are similar to the Hanford formation: sands, gravels, and cobbles, with varying fractions of silt.

Perched water was noted during drilling of well N-35 at a depth of approximately 9 m (30 ft). Well N-35, located immediately adjacent to the 1325-N LWDF crib, was installed after effluent disposal to that unit had begun.

The vadose zone in the 100-N Area ranges in thickness from 0 to 21 m (0 to 70 ft), going from the river in the west to the eastern edge of the area. The thickness of the vadose zone beneath the 100-N Area RCRA sites is 18 to 21 m (60 to 70 ft). Soil moisture data for the 100-N Area are limited, and range from 1 to 3% for sediments not moistened by effluent disposal (Hartman 1992).

4.1.4.3 Unconfined Aquifer. The unconfined aquifer in the 100-N Area is approximately 12 to 15 m (40 to 50 ft) thick and is situated primarily in the upper part of the Ringold Formation. In some locations, overlying portions of the Hanford formation were also saturated where groundwater mounds were present. The base of the aquifer is believed to be a laterally continuous clay-rich unit containing a series of paleosols. Lithologies in this unit are believed to range from clay and silt to sand. Most of the wells in the 100-N Area were completed in the upper portion of the unconfined aquifer; therefore, the thickness of the clay-rich unit is not known precisely at all locations. It is also not known if this clay-rich unit fully isolates the

Figure 26. Location Map Showing 1991 Shoreline Sampling Locations (Peterson and Johnson 1992).



unconfined aquifer from the underlying hydrologic units. There may be hydraulic communication between water-bearing units in the Ringold Formation (Hartman 1992a).

4.2 HYDROLOGIC RESPONSES TO EFFLUENT DISPOSAL

During peak reactor operations effluent flow to the 1325-N LWDF was continuous at a rate of approximately 3,785 to 6,057 L/min (1,000 to 1,600 gal/min) and could increase to 7,570 L/min (2,000 gal/min) during specific operations. Sources from other reactor support facilities were more intermittent (see Table 1). Over its period of operation, the 1325-N LWDF has had a significant effect on the water table in its vicinity. Figures 27, 28, 29, and 30 show changes in the water table between June 1988 and January 1993. Over the years there have been mounds under many of the effluent disposal sites. In Figures 27 through 30, there is evidence of mounding from the 1324-N/NA, 1325-N LWDF, and potentially the backwash lake. Figure 31 shows a conceptualization of groundwater flow in the 100-N Area during active discharging of liquid effluents. The active discharge unit shown in Figure 27 is an area of groundwater recharge; mounding creates a vertical component of water movement beneath the site. On a local scale, groundwater flows radially outward from the groundwater mound. However, the overall direction of flow in the uppermost aquifer is toward the Columbia River, which is characteristic of areas of groundwater discharge (Hartman 1993).

The groundwater appears to be reverting back to a natural/preHanford Site flow regime. Several factors illustrate this effect:

- Discharge to the 1325-N LWDF has not occurred since April 1991
- Recharge to the unconfined aquifer from 1989 to 1991 was minimal compared to the effects from changes in river stage
- Since June 1989, water levels beneath the 100-N Area have dropped over 6 m (20 ft) and the groundwater mound beneath 1325-N LWDF declined and dissipated
- The groundwater gradient has decreased from 0.01 in June of 1989 to 0.001-0.002 in January of 1993.

4.3 GROUNDWATER QUALITY

In general, parameters of concern in the 100 Areas are gross alpha, gross beta, tritium, nitrate, and chromium. Plume maps have been generated for all of these contaminants (Hartman and Peterson 1992). These contaminants are highly mobile in groundwater and were present in waste liquids and reactor coolant released in the 100 Areas. There is also evidence of tritium migrating from the 200 Areas northward through Gable Gap and towards the river. Several wells in the Gable Gap area and northward towards 100-K Area show elevated concentrations of tritium (Hartman and Peterson 1992).

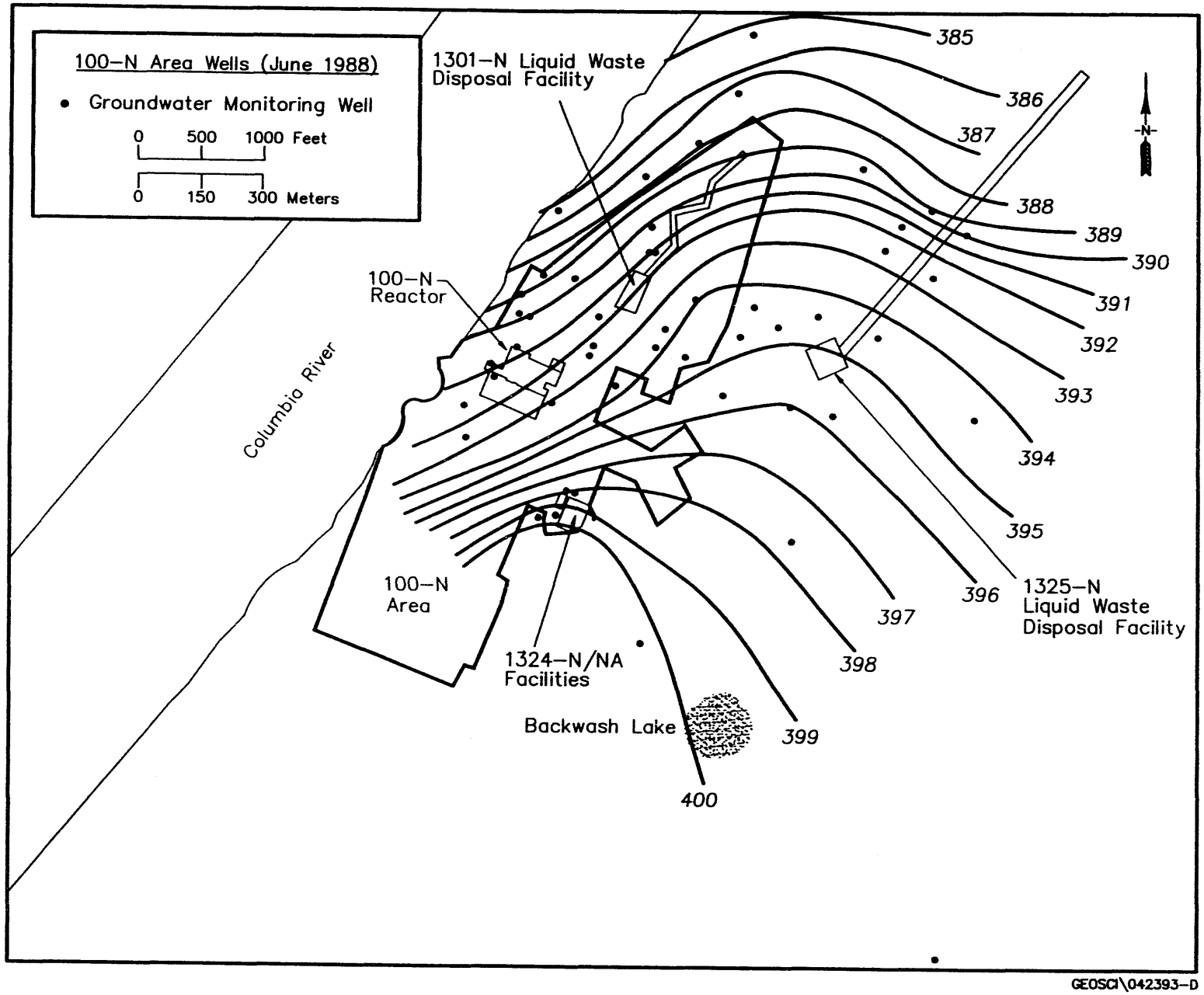


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Figure 27. Water Table in the 100-N Area and Vicinity, June 1988.

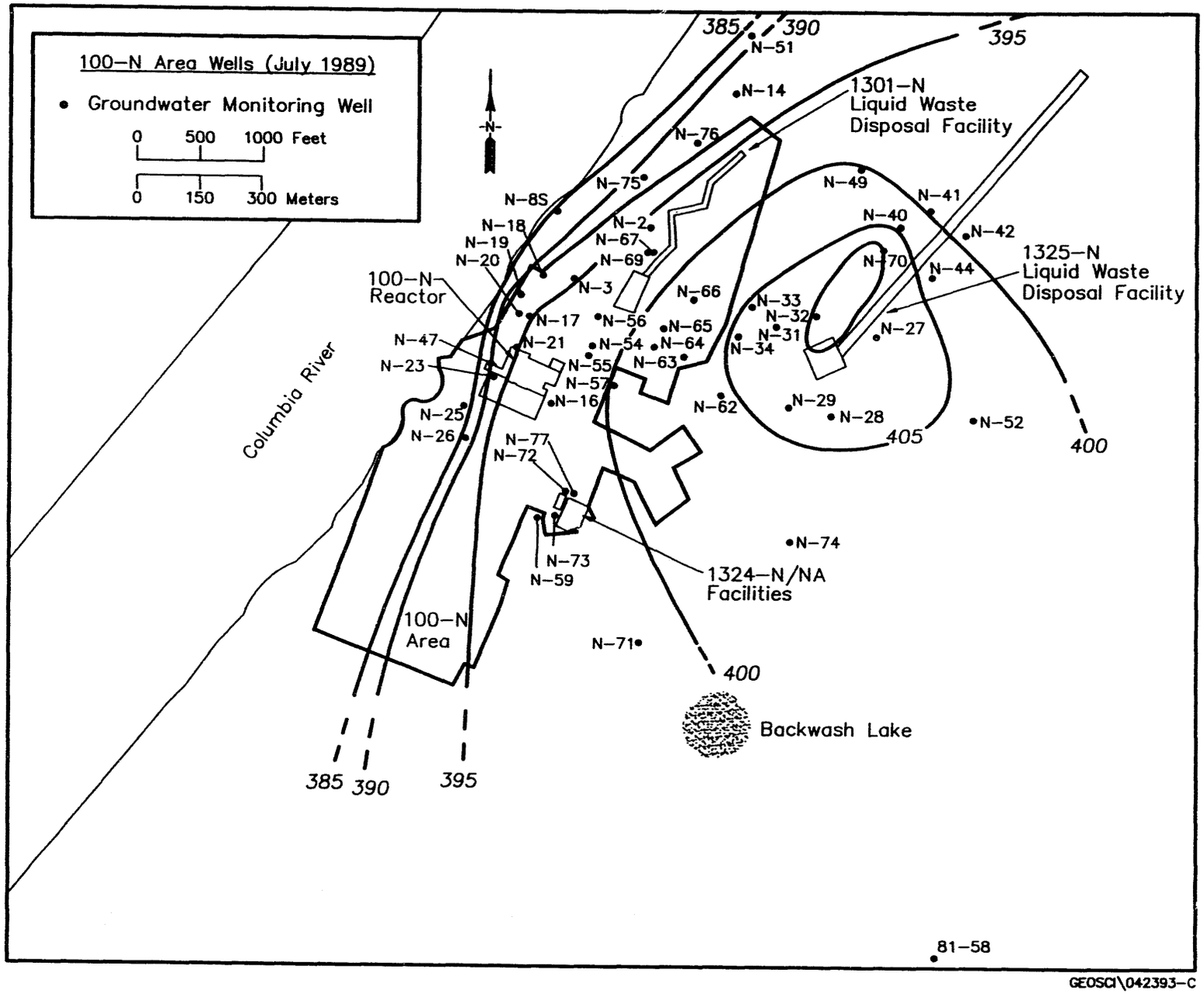


Figure 28. Water Table in the 100-N Area and Vicinity, July 1989.

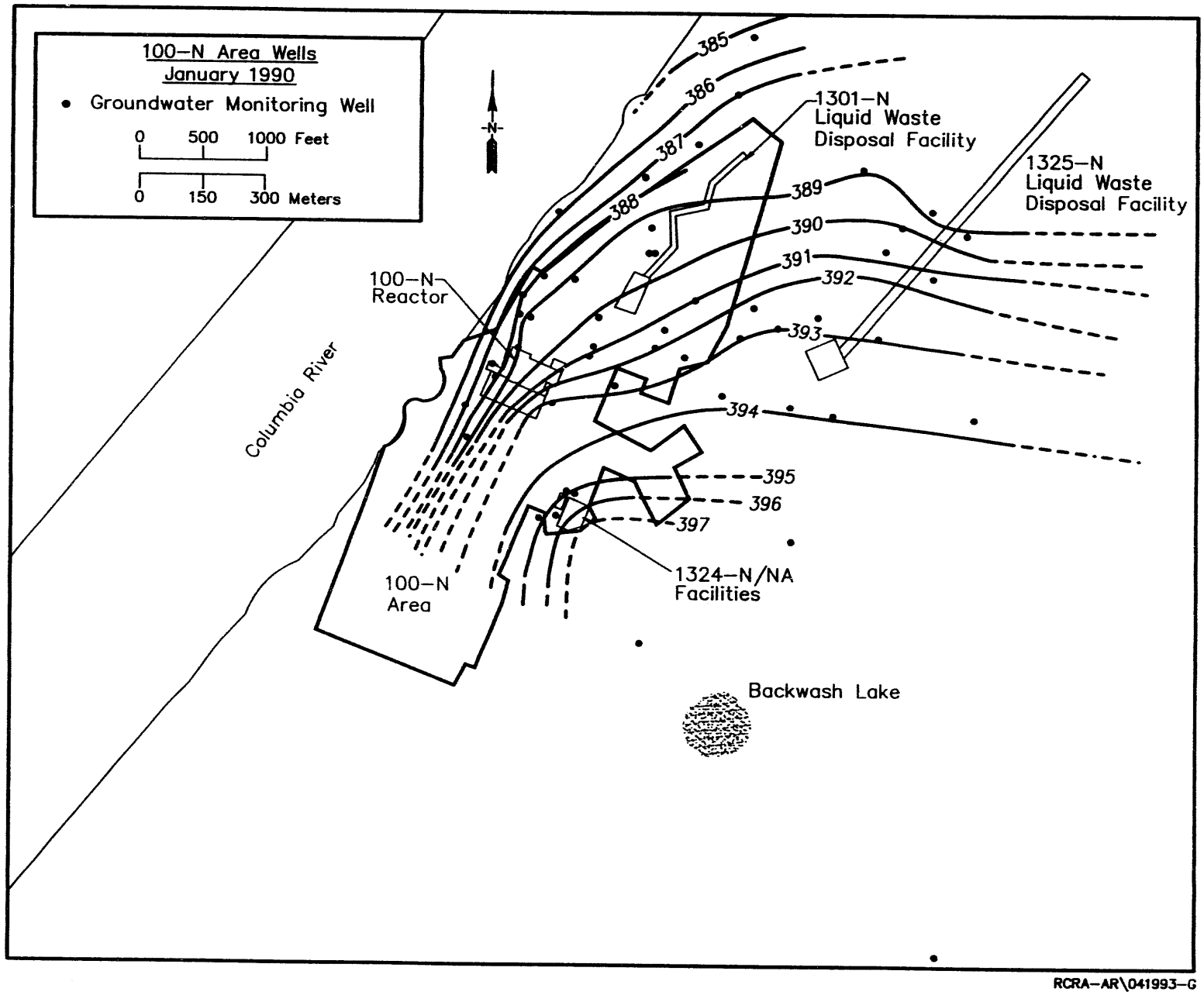


Figure 29. Water Table in the 100-N Area and Vicinity, January 1990.

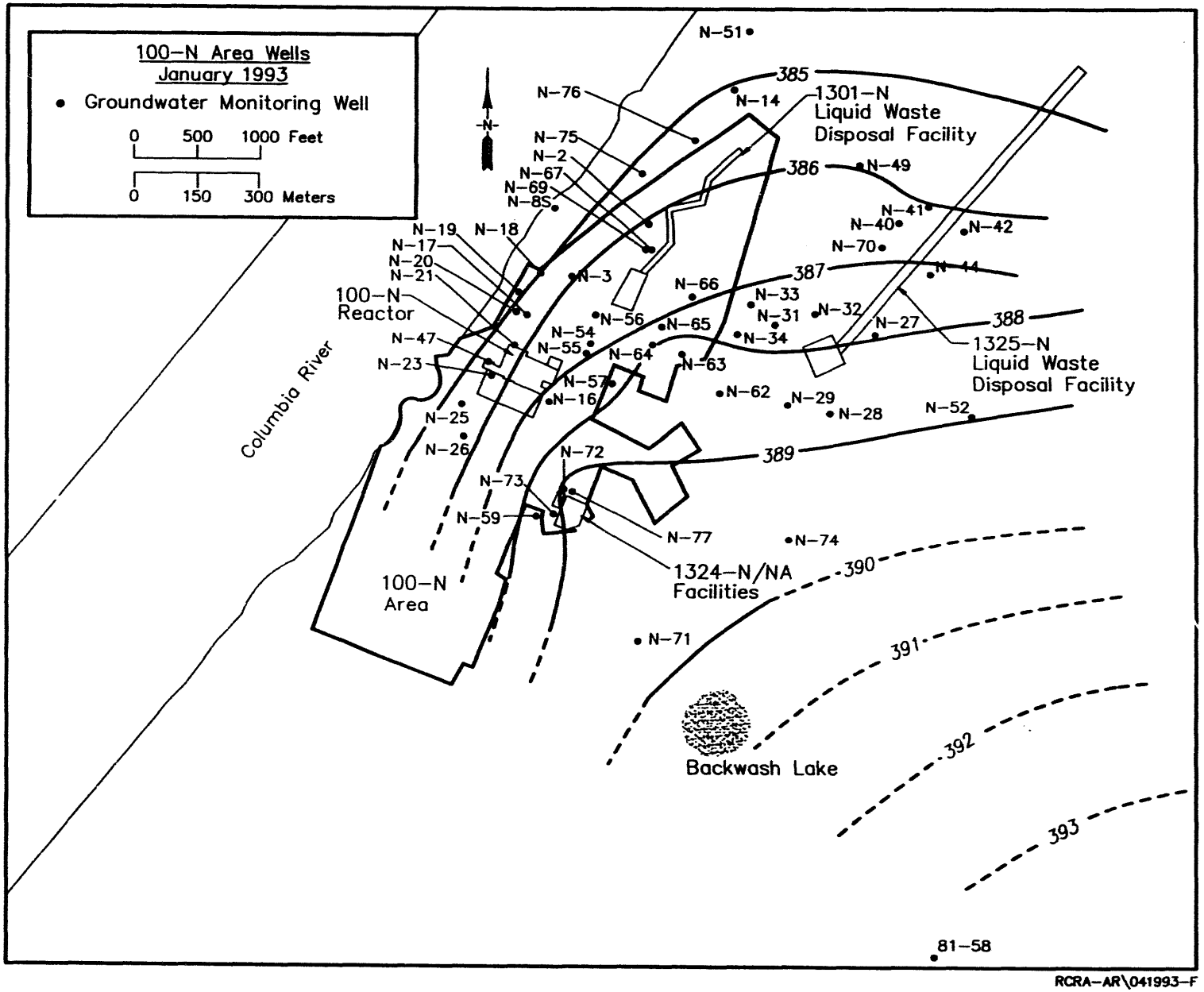
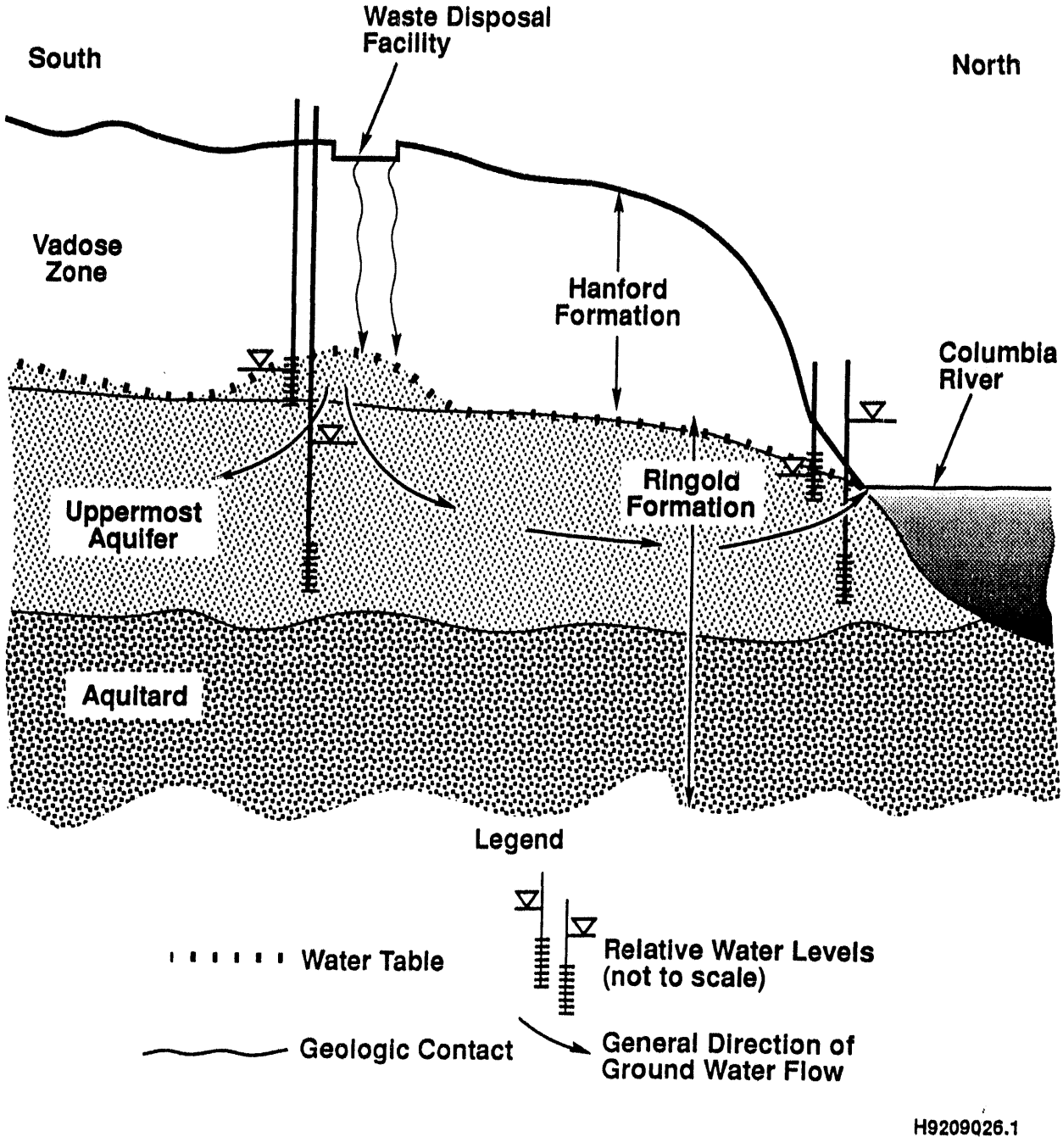


Figure 30. Water Table in the 100-N Area and Vicinity, January 1993.

Figure 31. Conceptual Drawing of Groundwater Flow in the Uppermost Aquifer at the 100-N Area (Hartman 1993).



Hartman and Peterson (1992) present interpretations of groundwater chemistry for the 100-N Area based on data collected in 1990. At that time, groundwater beneath most of the 100-N Area appeared to be contaminated by gross beta and tritium. Gross alpha was below the detection limit in the sampled wells. Nitrate was elevated above background in a few isolated wells, but below the drinking water standard. Chromium was less than or at the detection limit in the sampled wells.

The RCRA Groundwater Monitoring Programs for the 1301-N LWDF, the 1325-N LWDF, and the 1324-N/NA Facilities have identified the primary contaminants present in the groundwater in the 100-N Area. Monitoring data are reported in the RCRA Quarterly Reports. Interpretations of the data are presented in the RCRA Annual Reports. This report relies on the two most recent annual reports (DOE-RL 1992a, DOE-RL 1993).

Hydrochemical results for the groundwater monitoring wells at the 100-N Area were compared to Hanford sitewide natural background threshold values (BTV)-95% confidence interval. This was to determine if the 100-N Area and the surrounding area unconfined aquifer differed in chemistry from the sitewide background values for the unconfined aquifer. The following assumptions apply to the data set used in this analysis.

(1) Data include RCRA, CERCLA, and Operational Programs sampling data that had been entered into the Hanford Environmental Information System (HEIS) database up to mid-August 1993.

(2) All data are from samples collected in the 1990's.

(3) The data given for a year were averaged for the year and are rounded to the nearest whole number.

(4) All data are for unfiltered samples; except for the metals, for which both filtered and unfiltered results were examined.

The results of this comparison are as follows:

- Groundwater Quality Parameters--pH values are all within the 7 to 8 range, with the exception of well N-73. This particular well averages pH values near 9. This well exceeds the WWQS for pH (6.5 to 8.5). It is immediately downgradient of the 1324-N/NA Facility, which is probably the cause of the elevated pH values. Past-practice disposal activities, such as the neutralization of regeneration solutions, may have contributed excess alkaline solution or metal hydroxide precipitate (e.g., $CaOH_2 \rightarrow CaOH^-$) to the immediate area.

  Specific conductance concentrations range from 165 to 1400 μmhos/cm. Approximately half of the wells have concentrations above the site BTV of 539 μmhos/cm (Table 7). Concentrations that are elevated occur in wells N-3, N-17, N-18, N-20, N-21, N-25, N-26, N-47, N-54, N-55, N-56, N-57, N-58, N-59, N-60, N-61, and N-73. These elevated values may be attributed to several known contaminant sources in the vicinity of the wells (see Figure 24). A plume map for specific conductance is shown in Figure 32.

Table 7. Summary of Provisional Hanford Site Groundwater Background Values[a] (Johnson 1993). (3 sheets)

Constituent (concentration)	PNL results[b]	USGS results[b] (sample size)	WHC unconfined[b] (sample size)	WHC provisional threshold values
Aluminum (ppb)	<2	110 ± 139 (12)	<200 (50)	<200
Ammonium (ppb)	<50	NA	<50 (18)	<120
Arsenic (ppb)	3.9 ± 2.4	6.7 ± 3.7 (7)	<5 (14)	10
Barium (ppb)	42 ± 20	53 ± 14 (11)	41 ± 20 (53)	68.5
Beryllium (ppb)	<0.3	NA	<5 (16)	<5
Bismuth (ppb)	<0.02	NA	<5 (4)	<5
Boron (ppb)	<50	<50 (14)	<100 (35)	<100
Cadmium (ppb)	<0.2	<10 (1)	<10 (16)	<10
Calcium (ppb)	40,400 ± 10,300	40,857 ± 8,282 (14)	38,542 ± 11,023 (53)	63,600
Chloride-low (ppb)	NA	5,825 ± 1,355 (8)	5,032 ± 1,774 (53)	8,690
Chloride-high (ppb)	NA	20,667 ± 2,503 (6)	23,296 ± 2,463 (14)	28,500
Chloride-all (ppb)	10,300 ± 6,500	12,186 ± 7,842 (14)	8,848 ± 7,723 (67)	NC
Chromium (ppb)	4 ± 2	<50 (11)	<30 (8)	<30
Copper (ppb)	<1	<10 (10)	<30 (50)	<30
Fluoride (ppb)	370 ± 100	550 ± 330 (14)	437 ± 131[c] (47)	1,340 775[c]
Iron-low (ppb)	NA	22 ± 16[c]	<50 (34)	86
Iron-mid (ppb)	NA	NA	115 ± 52 (7)	291
Iron-high (ppb)	NA	NA	494 ± 118 (12)	818
Iron-all (ppb)	NA	NA	149 ± 199 (53)	NC
Lead (ppb)	<0.5	<30[c] (6)	<5 (15)	<5
Magnesium (ppb)	11,800 ± 3,400	10,814 ± 1,813 (14)	11,190 ± 2,578 (14)	16,480
Manganese-low (ppb)	NA	26 ± 27 (8)	<20 (33)	24.5
Manganese-high (ppb)	NA	150 ± 87 (3)	118 ± 17 (20)	163.5
Manganese-all (ppb)	--	60 ± 73 (11)	50 ± 55 (53)	NC

Table 7. Summary of Provisional Hanford Site Groundwater Background Values[a] (Johnson 1993). (3 sheets)

Constituent (concentration)	PNL results[b]	USGS results[b] (sample size)	WHC unconfined[b] (sample size)	WHC provisional threshold values
Mercury (ppb)	--	NA	<0.1 (14)	<0.1
Nickel (ppb)	<4	<50 (14)	<30 (23)	<30
Nitrate (ppb)	NA	3,224 ± 3,380 (13)	5,170 ± 3,576 (78)	12,400
Phosphate (ppb)	<1,000	140 ± 62 (3)	<1,000	<1,000
Potassium (ppb)	4,950 ± 1,240	5,900 ± 1,253 (14)	4,993 ± 1,453 (53)	7.975
Selenium (ppb)	<2	NA	<5 (14)	<5
Silver (ppb)	<10	NA	<10	<10
Silicon (ppb)	NA	16,786 ± 3,683 (14)	18,152 ± 4,974 (35)	26,500
Sodium (ppb)	18,260 ± 10.150	20,286 ± 7,907 (14)	15,774 ± 6,784 (53)	33,500
Strontium (ppb)	236 ± 102	159 ± 78 (14)	164 ± 47 (43)	264.1
Sulfate (ppb)	34,300 ± 16,900	41,286 ± 27,880 (14)	30,605 ± 22,611 (67)	90,500
Uranium (pCi/L)	1.7 ± 0.8	NA	1.7 ± 1.2 (10)	3.43
Vanadium (ppb)	17 ± 9	NA	9 ± 4 (18)	15
Zinc-low (ppb)	NA	14 ± 20 (11)	<50 (36)	<50
Zinc-high (ppb)	NA	373 ± 284 (3)	247 ± 165 (17)	673
Zinc-all (ppb)	6 ± 2	91 ± 190 (14)	95 ± 140 (53)	NC
Field alkaline (ppb)	NA	134,100 ± 20,469 (10)	137,758 ± 33,656 (31)	215,000
Lab alkaline (ppb)	123,000 ± 21,000	130,000 ± 8,165 (4)	133,717 ± 29,399 (52)	210,000
Field pH	NA	NA	7.57 ± 0.29 (57)	[6.90, 8.24]
Lab pH	7.64 ± 0.16	NA	7.75 ± 0.21 (52)	[7.25, 8.25]
TOC (ppb)	586 ± 347	NA	519 ± 367[c] (62)	2,610 1,610[c]
Field cond. (µmhos/cm)	NA	NA	344 ± 83 (22)	539

Table 7. Summary of Provisional Hanford Site Groundwater Background Values[a] (Johnson 1993). (3 sheets)

Constituent (concentration)	PNL results[b]	USGS results[b] (sample size)	WHC unconfined[b] (sample size)	WHC provisional threshold values
Lab cond. (µmhos/cm)	380 ± 82	NA	332 ± 93 (36)	530
TOX, LDL (ppb)	NA	NA	<20[c] (14)	60.8 37.6[c]
TOC (ppb)	NA	NA	31,772 ± 7,022 (48)	50,100
Gross alpha (pCi/L)	2.5 ± 1.4	NA	2.5 ± 1.5[c] (36)	63 5.79[c]
Gross beta (pCi/L)	19 ± 12	NA	7.1 ± 2.6[c] (44)	35.5 12.62[c]
Radium (pCi/L)	<0.2	NA	ND (10)	0.23

[a]Source: From Tables 5-9 and 5-11 of DOE-RL (1992b).
[b]Results shown are mean ± one standard deviation.
[c]Potential outlier observation(s) were removed.
LDL = lower detection limit.
NA = not available.
NC = not calculated.
ND = not detected.
PNL = Pacific Northwest Laboratory.
ppb = parts per billion.
TOC = total organic carbon.
TOX = total organic halogen.
USGS = United States Geological Survey.
WHC = Westinghouse Hanford Company.

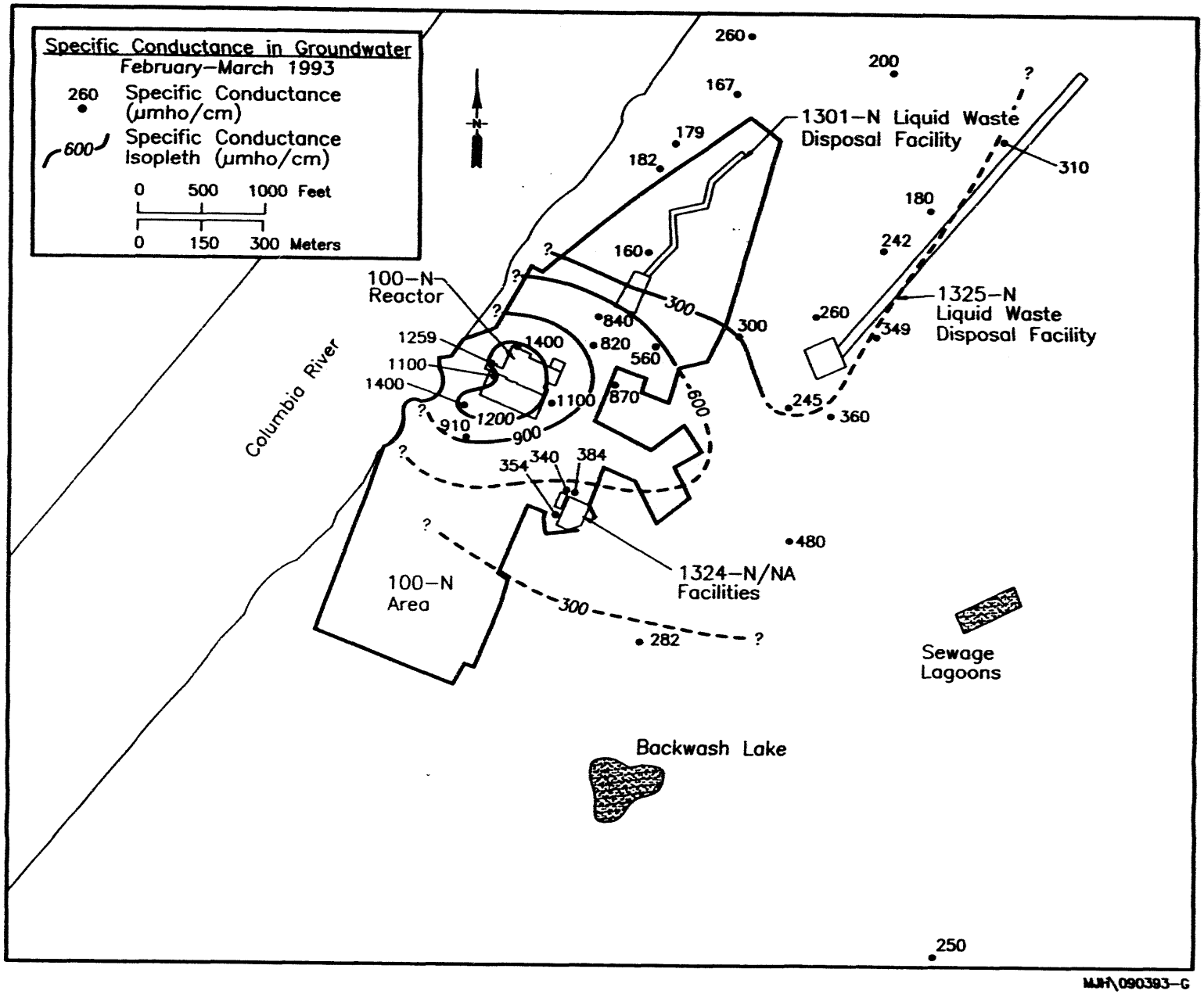


Figure 32. Distribution of Specific Conductance in Groundwater in the 100-N Area.

Alkalinity concentrations in most of the area are below the BTV (215 ppm) for the site unconfined aquifer (see Table 7). Only one well (N-17), has a concentration that exceeds this (350 ppm). This well is located to the north of the 105-N/109-N Facility (see Figure 24). The cause of this high alkalinity value is unknown.

Total organic carbon is below the site BTV for the unconfined aquifer (1,610 parts per billion [ppb]) in all of the wells in the 100-N Area (see Table 9).

Turbidity values in the majority of the wells are all near normal values (<10 nephelometric turbidity unit). There are a few wells with values between 10 to 25 nephelometric turbidity unit (N-3, N-16, N-26, and N-34). The first three wells are near the reactor and support facilities, and the last is downgradient of the 1325-N LWDF (see Figure 24). Three wells have values that range from 46 to 200 nephelometric turbidity unit (N-25, N-41, and N-64). These wells are located southwest of the 105-N/109-N Facilities, northwest of the 1325-N LWDF trench, and southeast of the 1301-N LWDF crib, respectively. The cause of these higher turbidity values is unknown.

- Cation concentrations--Calcium concentrations ranged from 2,100 to 130,000 ppb. The site unconfined aquifer BTV is 63,600 ppb (see Table 7). Less than half of the wells in the 100-N Area have values in excess of this BTV. There appears to be little difference between the unfiltered and filtered sample results. The majority of the wells with elevated concentrations are in the vicinity of the 105-N/109-N Facilities and 1301-N LWDF (see Figure 24).

  Sodium concentrations ranged from 2,800 to 220,000 ppb in the 100-N Area. The site BTV is approximately 33,500 ppb for the unconfined aquifer (see Table 7). The results for filtered and unfiltered samples are very similar. The highest concentrations are found in wells N-16 (175,00 ppb), N-21 (180,000 ppb), N-23 (15,000 ppb), N-25 (220,000 ppb), N-26 (108,500 ppb), N-47 (135,000 ppb), N-54 (40,000 ppb), N-57 (73,000 ppb), N-71 (41,500 ppb), and N-73 (72,000 ppb), downgradient of the 1324-N/NA Facility (see Figure 24). The elevated sodium is a result of past-practice disposal activities at this facility. A plume map for sodium is shown in Figure 33.

  Potassium concentrations ranged from 960 to 8,000 ppb. The BTV for the unconfined aquifer onsite is approximately 7,975 ppb (see Table 7). Only one well (N-21) (8,000 ppb) slightly exceeds this value.

  Ammonium concentrations ranged from at or below the contractually required quantification limit (CRQL) of 100 to 700 ppb. The site BTV for the unconfined aquifer is less than 120 ppb (see Table 7). Only seven wells show concentrations above 120 ppb. They are N-3 (700 ppb), N-25 (250 ppb), N-28 (400 ppb), N-64 (300 ppb), N-69 (150 ppb), N-70 (200 ppb), and N-73 (150 ppb). These wells are located near several facilities (see Figure 24), making it difficult to draw specific conclusions as to a possible source.

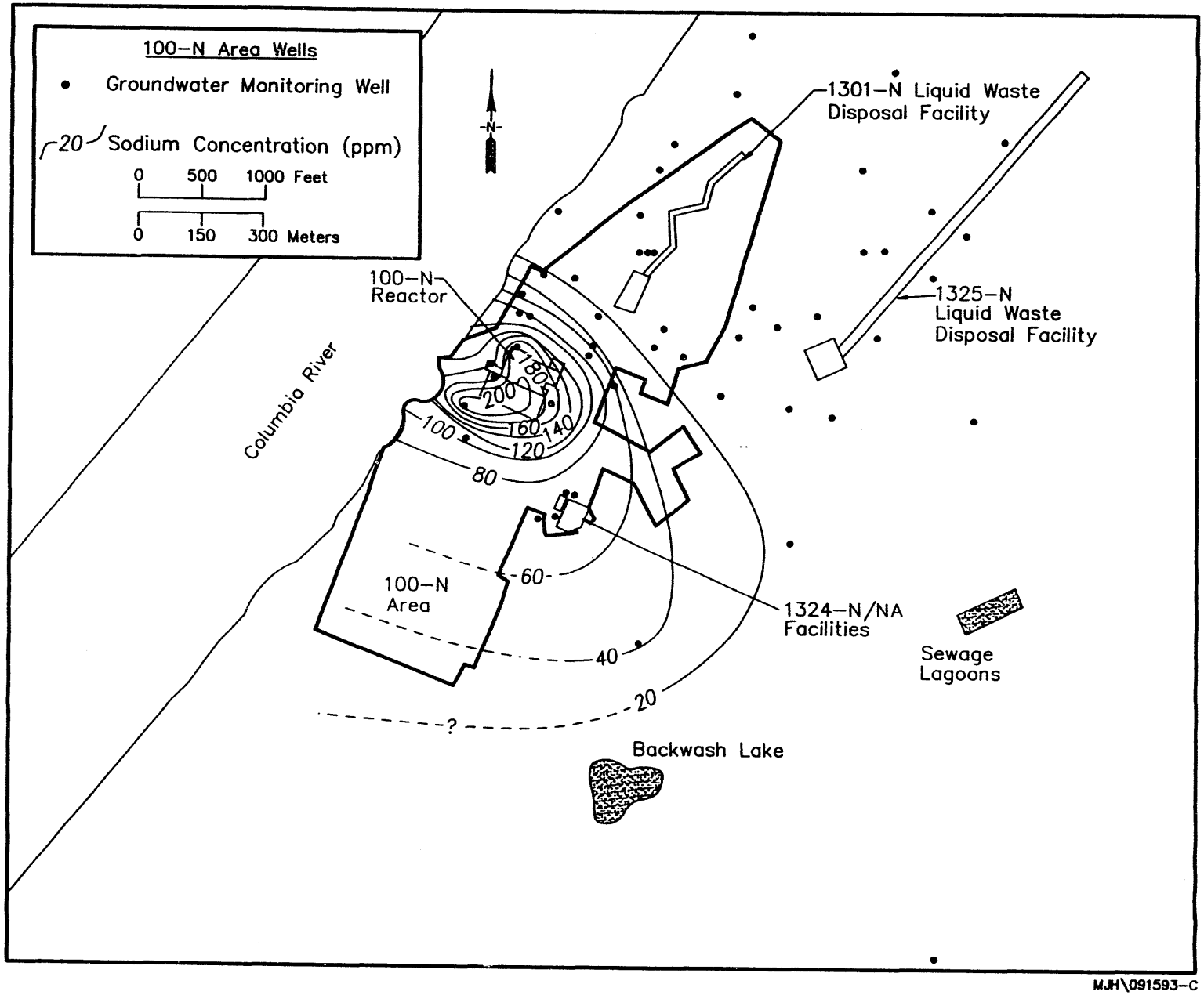


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Figure 33. Distribution of Sodium in Groundwater in the 100-N Area.

- Anion concentrations--Chloride concentrations ranged from 1,200 to 79,500 ppb for wells in the 100-N Area. The BTV value for the site unconfined aquifer (see Table 7) ranges from 8,690 to 28,500 ppb (because of two populations of chloride). Well N-16 has a concentration of 79,500 ppb. Eight wells have concentrations greater than 10,000 ppb: N-3 (16,000 ppb), N-17 (17,000 ppb), N-21 (14,500 ppb), N-26 (14,500 ppb), N-47 (11,400 ppb), N-54 (13,000 ppb), N-57 (16,000 ppb), and N-64 (12,000 ppb). These eight wells are downgradient of the 1324-N/NA Facility (see Figure 24). The elevated chloride in the unconfined aquifer in 100-N Area is a result of past-practice disposal activities at the 1324-N/NA Facility. None of the wells exceeds the WWQS for chloride, which is 250,000 ppb. A plume map for chloride is shown in Figure 34.

  Fluoride concentrations ranged from at or below the CRQL of 100 to 1,300 ppb. The site BTV for the unconfined aquifer is 775 ppb (see Table 7). Seven wells are higher than the site BTV: N-17 (1,300 ppb), N-21 (1,250 ppb), N-23 (1,000 ppb), N-25 (1,300 ppb), N-26 (1,100 ppb), N-47 (1,100 ppb), and N-73 (1,150 ppb). These wells are located near several potential sources (see Figure 24).

  Nitrate concentrations ranged from 900 to 50,000 ppb in the 100-N Area. The site BTV for the unconfined aquifer is approximately 12,400 ppb (see Table 7). Nine wells have concentrations in excess of this value: N-3 (20,000 ppb), N-16 (22,000 ppb), N-21 (21,000 ppb), N-26 (50,000 ppb), N-32 (31,500 ppb), N-54 (15,000 ppb), N-56 (16,500 ppb), N-64 (46,500 ppb), and N-70 (13,400 ppb). The WWQS for nitrate as nitrogen is 10,000 ppb. All but 11 of the wells sampled exceed this value. The majority of the wells with elevated concentrations are located near past-practice disposal sites, but it is difficult to determine the source of the nitrate (see Figure 24). A plume map for nitrate is shown in Figure 35. Sewage effluent disposal to the ground may be a source of nitrate contamination. Another possibility is a chemical used for decontamination in the 100-N Area. Diethylthiourea is mixed with phosphoric acid for decontamination and contains amine groups ($C_2H_5NHCSNHC_2H_5$). Nitrate concentrations that were higher than the surrounding 600 Area groundwaters were noted by Hartman and Peterson (1992) in their paper summarizing the hydrology of the northern Hanford Site.

  Nitrite concentrations were at or below the CRQL of 200 ppb for all but one well in the 100-N Area. Well N-3 had a concentration of 850 ppb.

Figure 34. Distribution of Chloride in Groundwater in the 100-N Area.

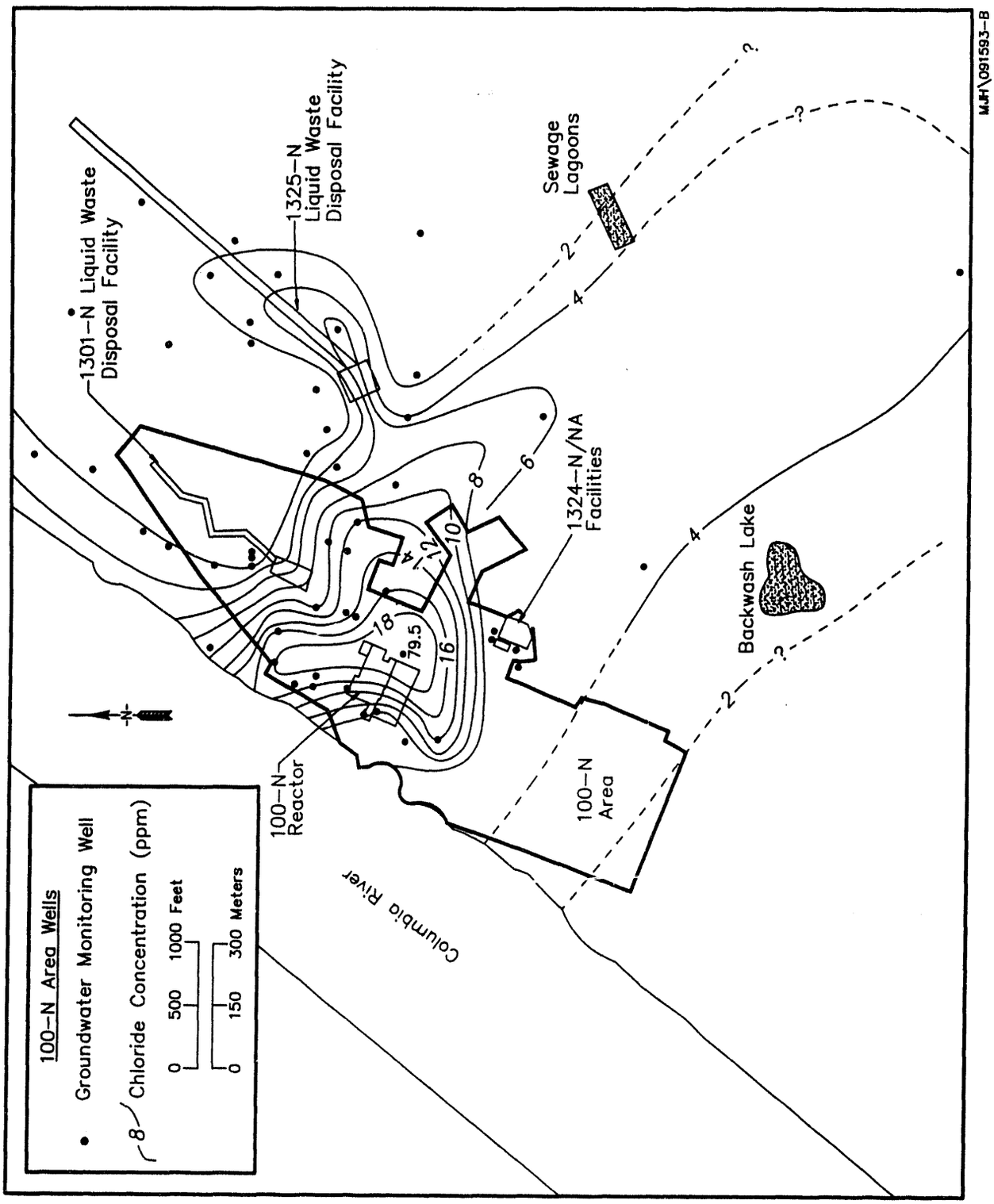
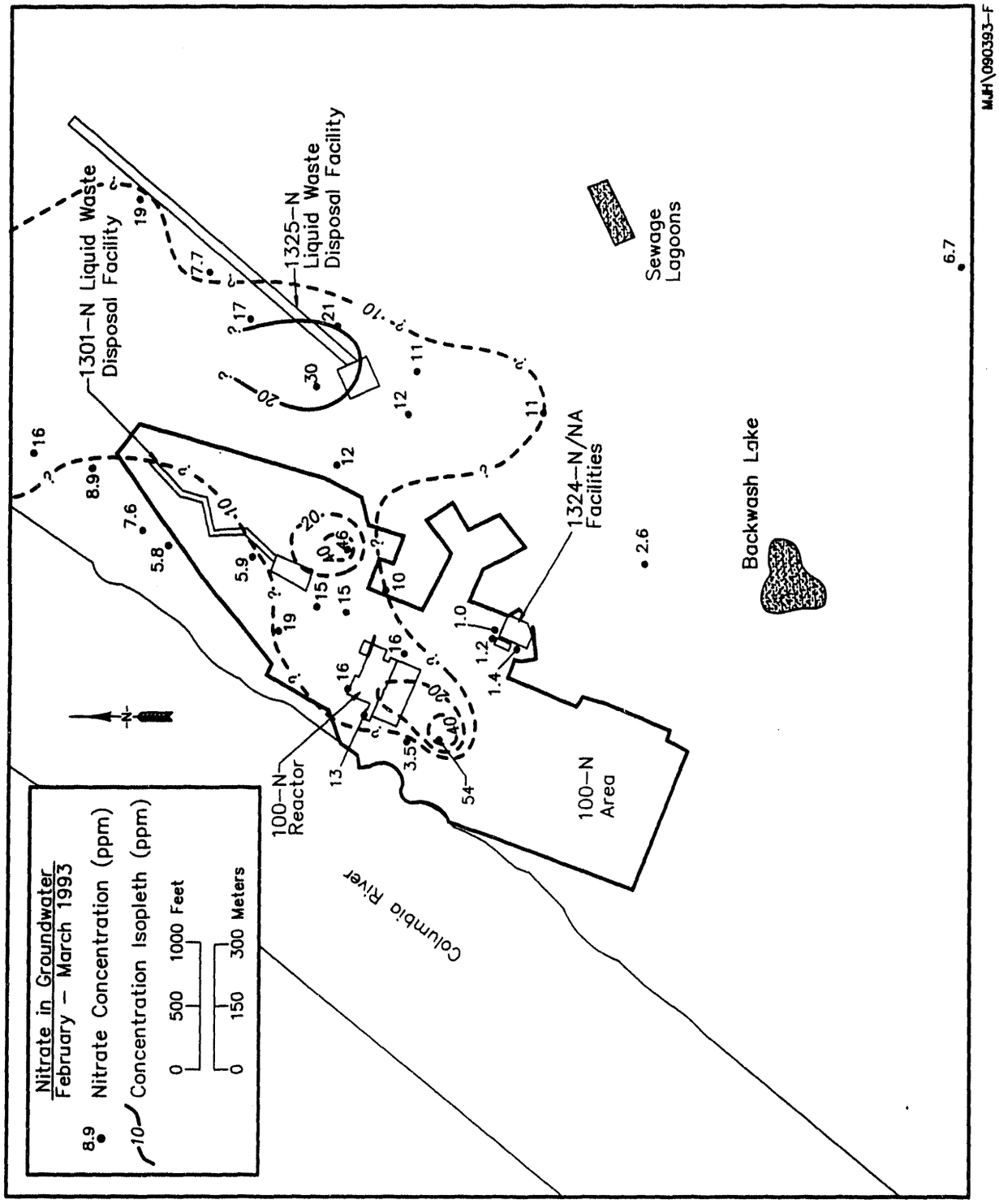


Figure 35. Distribution of Nitrate in Groundwater in the 100-N Area.



Sulfate concentrations ranged from 12,500 to 550,000 ppb. The site BTV for sulfate is around 90,500 ppb (see Table 7). Of the 42 well samples, 33 have concentrations over 50,000 ppb. Fourteen of these wells have concentrations in the site BTV. There is a known sulfate plume in the 100-N Area from past-practice disposal activities. The 1325-N/NA Facility is the source of the sulfate in the 100-N Area. Eight of the high sulfate concentration wells also exceed the WWQS for sulfate, which is 250,000 ppb. These wells are N-17 (350,000 ppb), N-21 (520,000 ppb), N-23 (400,000 ppb), N-25 (550,000 ppb), N-47 (440,000 ppb), N-54 (270,000 ppb), N-56 (255,000 ppb), and N-57 (300,000 ppb) (see Figure 24). A plume map for sulfate is shown in Figure 36.

Phosphate concentrations were at or below the CRQL of 400 ppb for all but one well in the 100-N Area. Well N-73 had a concentration of 500 ppb.

Bromide concentrations were at or less than the CRQL of 500 ppb (see Table 7).

- Metal concentrations--Aluminum concentrations ranged from 53 to 8,300 ppb. All but two of the values were below the site BTV (<200 ppb) for the unconfined aquifer (see Table 7). The two wells that exceeded this value are N-57 (8,300 ppb) and N-64 (1,400 ppb). These wells are located fairly close to each other, south of the 1301-N LWDF (see Figure 24). The source of elevated aluminum values is unknown.

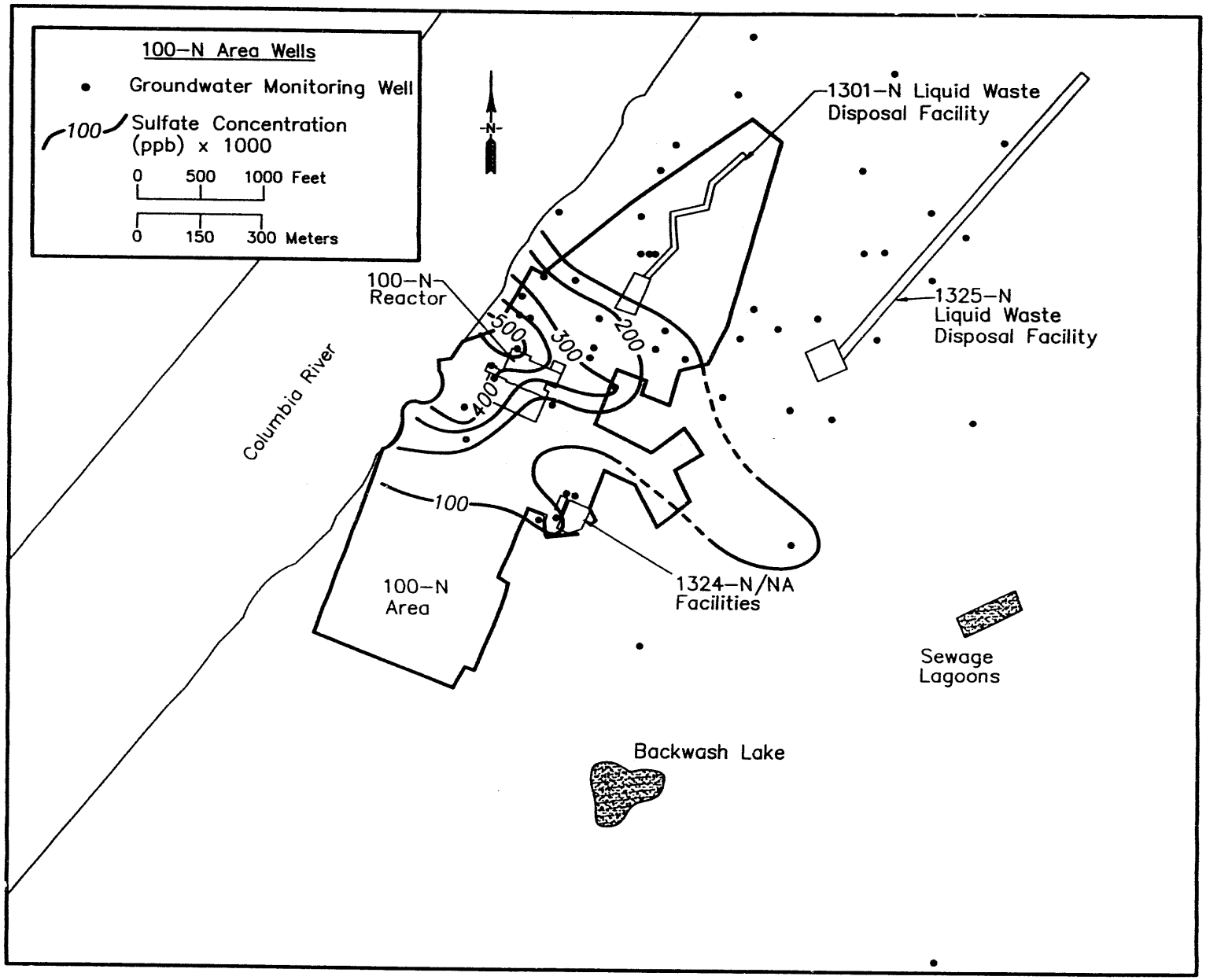
  Arsenic concentrations ranged from 4 to 17 ppb. The site BTV for the unconfined aquifer is 10 ppb (see Table 7). Three wells exceeded this value: N-34 (13 ppb), N-57 (17 ppb), and N-73 (13 ppb). The wells are not adjacent to each other and there is no source of arsenic identified (see Figure 24).

  Barium concentrations ranged from 17 to 225 ppb. The site BTV for the unconfined aquifer is 69 ppb (see Table 7). Six wells exceed this value: N-3 (71 ppb), N-16 (170 ppb), N-54 (70 ppb), N-56 (100 ppb), N-57 (225 ppb), and N-64 (123 ppb). None of the wells exceed the WWQS of 1,000 ppb. The wells that are elevated in barium are located south and east of the 1301-N LWDF (see Figure 24).

  Boron concentrations are all well below the site BTV (<100 ppb) (see Table 7).

  Chromium concentrations for unfiltered samples ranged from at or below the CRQL of 20 to 140 ppb. The site BTV for the unconfined aquifer is less than 30 ppb (see Table 7). Ten wells exceed the site BTV: N-16 (65 ppb), N-26 (65 ppb), N-34 (31 ppb), N-57 (95 ppb), N-64 (56 ppb), N-69 (38 ppb), N-71 (49 ppb), N-72 (34 ppb), N-73 (92 ppb), and N-74 (140 ppb) (see Figure 24).

Figure 36. Distribution of Sulfate in Groundwater in the 100-N Area.



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Filtered samples were all at or below the CRQL of 200 ppb. Six of the wells listed above exceed the WWQS for chromium, which is 50 ppb. There are chromium plumes in other 100 Areas, but none is identified in the 100-N Area presently. Elevated chromium concentrations in unfiltered samples from new wells may be attributed to fragments of stainless steel, produced during installation of well casings and screens.

Copper concentrations range from at or below the CRQL of 20 to 80 ppb. The site BTV concentration for the unconfined aquifer is less than 30 ppb (see Table 7). Only one well exceeds this value, N-57 (80 ppb) (see Figure 24).

Iron concentrations ranged from at or below the CRQL of 20 to 45,020 ppb. The site BTV concentration for the unconfined aquifer is 86 to 818 ppb (see Table 7). Nine wells in the 100-N Area exceed this value: N-3 (3,665 ppb), N-16 (9,500 ppb), N-25 (6,750 ppb), N-26 (3,550 ppb), N-34 (17,000 ppb), N-41 (21,000 ppb), N-57 (45,020 ppb), N-62 (950 ppb), and N-64 (7,433 ppb). The wells listed above, and the following wells all exceed the WWQS for iron, which is 300 ppb: N-14 (375 ppb), N-23 (550 ppb), N-73 (560 ppb), and N-74 (433 ppb) (see Figure 24). Filtered samples were all reduced in concentration compared to unfiltered samples. Iron concentrations can be elevated in unfiltered samples from new wells because of the influence of the stainless steel casing and well screen, or from iron precipitated on sediments pulled in through the well screen.

Lead concentrations were at or below the CRQL of 5 ppb, except for well N-57, which had a concentration of 14 ppb (see Figure 24). This value does not exceed the WWQS of 50 ppb.

Magnesium concentrations ranged from 370 to 23,000 ppb. The site BTV for the unconfined aquifer is 16,480 ppb (see Table 7). Seven of the wells had values which exceeded this BTV: N-21 (23,000 ppb), N-23 (17,000 ppb), N-25 (22,000 ppb), N-47 (20,000 ppb), N-56 (19,000 ppb), N-57 (20,000 ppb), and N-64 (16,667 ppb) (see Figure 24).

Manganese concentrations ranged from at or below the CRQL of 10 to 295 ppb. The site BTV for the unconfined aquifer is 25 to 164 ppb (because of multiple populations of background manganese) (see Table 7). Five wells exceed the ranges upper value: N-3 (177 ppb), N-16 (295 ppb), N-34 (180 ppb), N-57 (290 ppb), and N-64 (170 ppb) (see Figure 24). The wells listed above, and the following wells all exceed the WWQS for manganese, which is 50 ppb: N-25 (105 ppb), N-26 (70 ppb), and N-41 (100 ppb). Concentrations of filtered samples were lower than unfiltered sample concentrations.

Nickel concentrations ranged from at or below the CRQL of 30 to 93 ppb. The site BTV for the unconfined aquifer is less than 30 ppb (see Table 7). Five wells exceed the BTV value: N-57 (55 ppb), N-64 (93 ppb), N-71 (36 ppb), N-73 (42 ppb), and N-74 (59 ppb) (see Figure 24). Only three of the wells listed above exceed the WWQS

for nickel, which is 50 ppb. Filtered samples were all at or below the CRQL. Nickel concentrations can be elevated in unfiltered samples from new wells because of the influence of the stainless steel casing and well screen.

Strontium (nonradiological) concentrations ranged from 79 to 426 ppb. The site BTV for the unconfined aquifer is 264 ppb (see Table 7). Four wells exceed the BTV value: N-16 (317 ppb), N-21 (318 ppb), N-47 (283 ppb), and N-55 (426 ppb) (see Figure 24). Filtered and unfiltered samples were in the same range of concentrations.

Vanadium concentrations ranged from 20 to 110 ppb. The site BTV for the unconfined aquifer is 15 ppb (see Table 7). All the wells exceed this value. There is no WWQS for vanadium. The filtered samples were within a similar range of concentrations as the unfiltered samples, with the exception of well N-57 (see Figure 24). This well had a concentration of 110 ppb unfiltered and 30 ppb filtered. Most of the unfiltered samples were similar to this elevated unfiltered value. The cause of elevated vanadium concentrations in the groundwater at 100-N Area is unknown.

The following metals were equal or less than CRQLs: antimony (200 ppb), beryllium (3 ppb), cadmium (10 ppb), cobalt (20 ppb), lithium (10 ppb), molybdenum (40 ppb), selenium (10 ppb), silver (20 ppb), thallium (5 ppb), tin (100 ppb), titanium (60 ppb).

Mercury and uranium were not detected in the groundwater samples analyzed.

- Organic concentrations--Hydrazine concentrations all at or below CRQL of 30 ppb.

  All other organics analyzed were at or less than the CRQL for each constituent.

- Radionuclide concentrations--Except for tritium and strontium-90, the radionuclide data are at or below the limit of quantification for the following constituents: antimony-125, cobalt-60, ruthenium-106, and cesium-137 (Tables 8, 9, 10, and 11, respectively).

  Strontium-90 concentrations range from 0 to 1,120 pCi/L. The values above background range from 36 to 1,120 pCi/L in the following wells: N-2 (92 ppb), N-3 (706 ppb), N-14 (1,120 ppb), N-17 (220 ppb), N-27 (214 ppb), N-28 (112 ppb), N-29 (343 ppb), N-34 (55 ppb), N-54 (317 ppb), N-56 (227 ppb), and N-57 (36 ppb). These wells are located downgradient of the 1301-N and 1325-N LWDFs (see Figure 24). There is a known plume of strontium-90 contamination in the 100-N Areas (Figure 37). The elevated concentrations of strontium-90 result from past-practice disposal operations at the 1301-N and the 1325-N LWDFs.

Table 8. Antimony-125 Limit of Detection/Limit of Quantification Calculated from Field Blanks.[a]

Time period	Number of blanks	Blank mean (pCi/L)	Blank standard deviation (pCi/L)	LOD[b] (pCi/L)	LOQ[b] (pCi/L)
01/02/92 to 03/20/92	7	3.73	5.666	17.0	56.7
04/07/92 to 06/11/92	5	-4.57	5.526	16.6	55.3
07/02/92 to 09/11/92	7	3.93	7.194	21.6	71.9
11/05/92 to 11/10/92	2	10.00	5.664	17.0	56.6
01/02/92 to 11/10/92	21	2.42	6.218	18.7	62.2
02/25/93 to 05/11/93	5	5.66	14.433	43.3	144.3

[a]The field blanks are QC samples that are introduced into a process to monitor the performance of the system. The calculated LOD/LOQ (using field blanks) measure the errors in the entire measurement system.

[b]LOD is 3 times blank standard deviation. LOQ is 10 times blank standard deviation.

LOD = limit of detection.
LOQ = limit of quantification.
QC = quality control.

Table 9. Cobalt-60 Limit of Detection/Limit of Quantification Calculated from Field Blanks.[a]

Time period	Number of blanks	Blank mean (pCi/L)	Blank standard deviation (pCi/L)	LOD[b] (pCi/L)	LOQ[b] (pCi/L)
01/02/92 to 03/20/92	7	1.796	4.943	14.8	49.4
04/07/92 to 06/11/92	5	0.060	3.840	11.5	38.4
07/02/92 to 09/11/92	7	1.399	3.288	9.9	32.9
11/05/92 to 11/10/92	2	-5.730	3.521	10.6	35.2
01/02/92 to 11/10/92	21	0.533	4.079	12.2	40.8
02/05/93 to 05/11/93	5	-1.614	4.515	13.5	45.2

[a]The field blanks are QC samples that are introduced into a process to monitor the performance of the system. The calculated LOD/LOQ (using field blanks) measure the errors in the entire measurement system.

[b]LOD (limit of detection) is 3 times blank standard deviation. LOQ (limit of quantification) is 10 times blank standard deviation.

LOD = limit of detection.

LOQ = limit of quantification.

Table 10. Ruthenium-106 Limit of Detection/Limit of Quantification Calculated from Field Blanks.[a]

Time period	Number of blanks	Blank mean (pCi/L)	Blank standard deviation (pCi/L)	LOD[b] (pCi/L)	LOQ[b] (pCi/L)
01/02/92 to 03/20/92	7	12.24	20.641	61.9	206.4
04/07/92 to 06/11/92	5	2.10	36.288	108.9	362.9
07/02/92 to 09/11/92	7	-8.29	22.853	68.6	228.5
11/05/92 to 11/10/92	2	-20.30	8.202	24.6	82.0
01/02/92 to 11/10/92	21	-0.12	25.465	76.4	254.6
02/25/93 to 05/11/93	5	1.82	30.018	114.1	380.2

[a]The field blanks are QC samples that are introduced into a process to monitor the performance of the system. The calculated LOD/LOQ (using field blanks) measure the errors in the entire measurement system.

[b]LOD is 3 times blank standard deviation. LOQ is 10 times blank standard deviation.

LOD = limit of detection.
LOQ = limit of quantification.
QC = quality control.

Table 11. Cesium-137 Limit of Detection/Limit of Quantification Calculated from Field Blanks.[a]

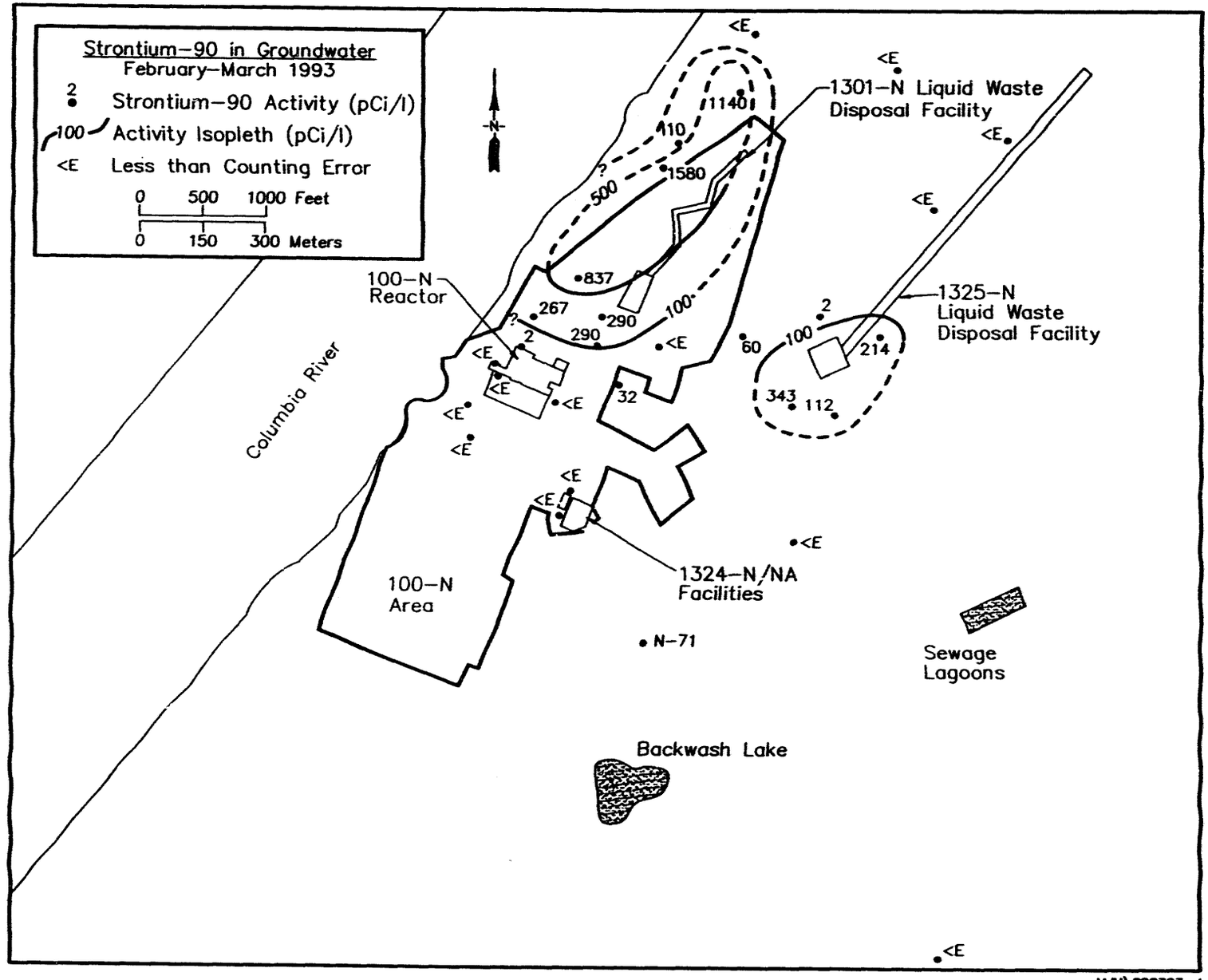
Time period	Number of blanks	Blank mean (pCi/L)	Blank standard deviation (pCi/L)	LOD[b] (pCi/L)	LOQ[b] (pCi/L)
01/02/92 to 03/20/92	7	-3.992	2.943	8.8	29.4
04/07/92 to 06/11/92	5	-0.987	4.857	14.6	48.6
07/02/92 to 09/11/92	7	1.149	3.382	10.1	33.8
11/05/92 to 11/10/92	2	1.866	2.269	6.8	22.7
01/02/92 to 11/10/92	21	-1.01	3.598	10.8	36.0
02/25/93 to 05/11/93	5	-1.43	1.728	5.2	17.3

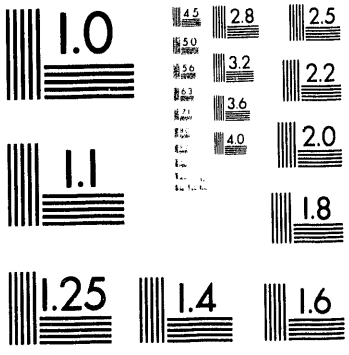
[a]The field blanks are QC samples that are introduced into a process to monitor the performance of the system. The calculated LOD/LOQ (using field blanks) measure the errors in the entire measurement system.

[b]LOD is 3 times blank standard deviation. LOQ is 10 times blank standard deviation.

LOD = limit of detection.
LOQ = limit of quantification.
QC = quality control.

Figure 37. Distribution of Strontium-90 in Groundwater in the 100-N Area.




1.0
4.5
2.8
2.5
5.0
3.2
2.2
5.6
6.3
3.6
7.1
4.0
2.0
1.1
1.8
1.25
1.4
1.6

2 of 2

Tritium concentrations range from 39 to 78,600 pCi/L. All but one value is above 100 pCi/L. The wells located downgradient of the 1301-N and 1325-N LWDFs have the highest concentrations of tritium (see Figure 24). Varying concentrations of tritium are detected in all the wells. There is a known plume of tritium contamination in the 100-N Areas (Figure 38). The elevated concentrations of tritium are caused by past-practice disposal operations at the 1301-N and 1325-N LWDFs.

4.4 SOIL COLUMN AND AQUIFER SEDIMENT CHEMICAL FACTORS

4.4.1 Soil Factors

The soil column or vadose zone beneath the 1325-N LWDF is composed of predominately sands and gravels, with minor fine-grained sediments (silts and loess). Sediments in the 100-N Area also contain 0.5 to 1.6% $CaCo_3$ (Hartman 1992). During disposal to the 1325-N LWDF, the sample analytical data indicated that effluent discharges contained numerous radionuclides in concentrations which, in most cases, greatly exceeded 1/25th of the derived concentration guides (Group A study guideline concentrations [WHC 1990a]). While concentrations of these constituents have decreased since the N Reactor was placed in dry layup, concentrations of some radioactive constituents remain high as a result of residual contamination in the soil column. Tritium and strontium-90 radionuclides have travelled through the soil column to the water table, and contaminated plumes have resulted. Chemical contaminants include chloride, sodium, calcium, strontium, nitrate, and sulfate. In addition, there appears to be elevated concentrations of metals in the groundwater, including iron, magnesium, chromium, and vanadium.

To estimate contaminant migration rates, the wastewater discharged to the crib is characterized as neutral to basic (pH = 7.8) and is low in both organic and inorganic constituents. Under these conditions, most of the cationic constituents tend to be immobile (e.g., calculated migration rates on the order of 1 to 3 cm/day). Tritium and the anionic constituents (iodine-131 and ruthenium-106 are treated as nonsorbing, for which a migration rate equivalent to water is assumed (approximately 360 cm/day [142 in./day] (WHC 1990a).

4.4.2 Aquifer Sediment Factors

The water softener regenerant from the 1324-N/NA Facility may have an effect on strontium-90 mobility and transport because of competing ion effects. This effect involves several competing divalent cations, caused by the presence of several alkali and alkali-earth metals removed from river water in the water softener process, and then discharged with the regenerant. The most common competing ion is Ca^{2+}, but other divalent cations will also compete with strontium-90 to a less degree (Cantrell and Serne 1993, see Appendix A).

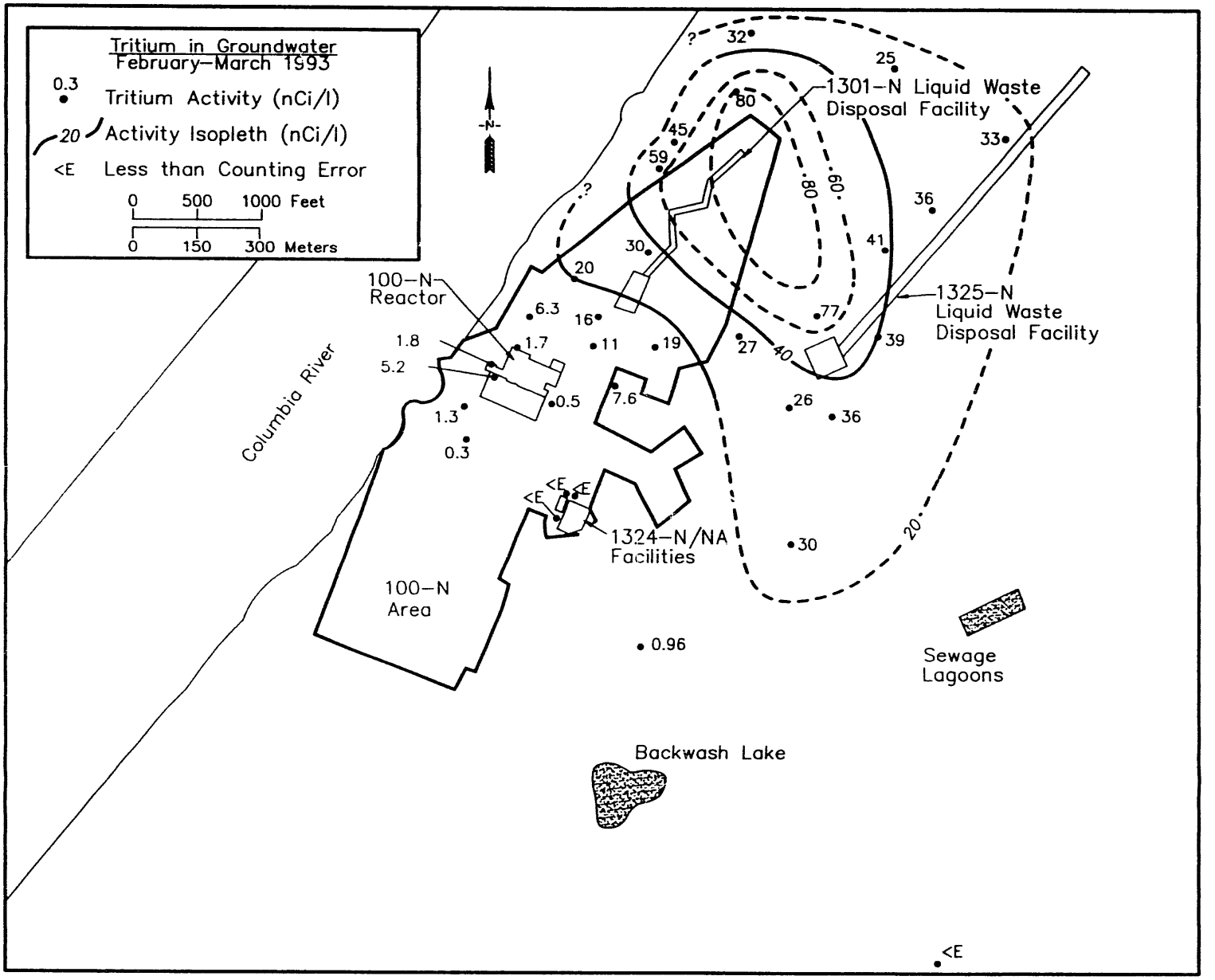


Figure 38. Distribution of Tritium in Groundwater in the 100-N Area.

4.5 SUMMARY OF CONCEPTUAL MODEL OF MOISTURE MOVEMENT AND CONTAMINANT TRANSPORT

A conceptual model of the hypothetical impact of occasional slug releases to the 1325-N Crib is summarized as follows (Figure 39):

(1) The principal contaminants of concern are the long-lived radionuclides strontium-90 and cesium-137, which were added to the crib at the rate of approximately 1 Ci/month during the 5-year period of use (1983 to 1988).

(2) The centroid of the strontium-90 and cesium-137 radionuclide inventory (approximately 60 Ci each, assuming no decay) reaches the water table near the end of the operating period.

(3) A (327,059-L [86,400-gal]) slug release of 1 month's accumulation at 8 L/min (2 gal/min) is discharged as a random event. Contaminants in the spent emergency water are assumed to be insignificant in comparison to the inventory already in the soil column beneath the crib. The spent emergency water is assumed to have a chemical composition similar to river water (low salt content; slightly basic pH; no surfactants, detergents, or complexants).

(4) The soil beneath the crib has been previously wetted, so there is very little additional holding capacity for retention of the hypothetical slug release. The slug release displaces 327,059 L (86,400 gal) of water from pore fluid in the soil column assumed to be in equilibrium with the adsorbed strontium-90 and cesium-137.

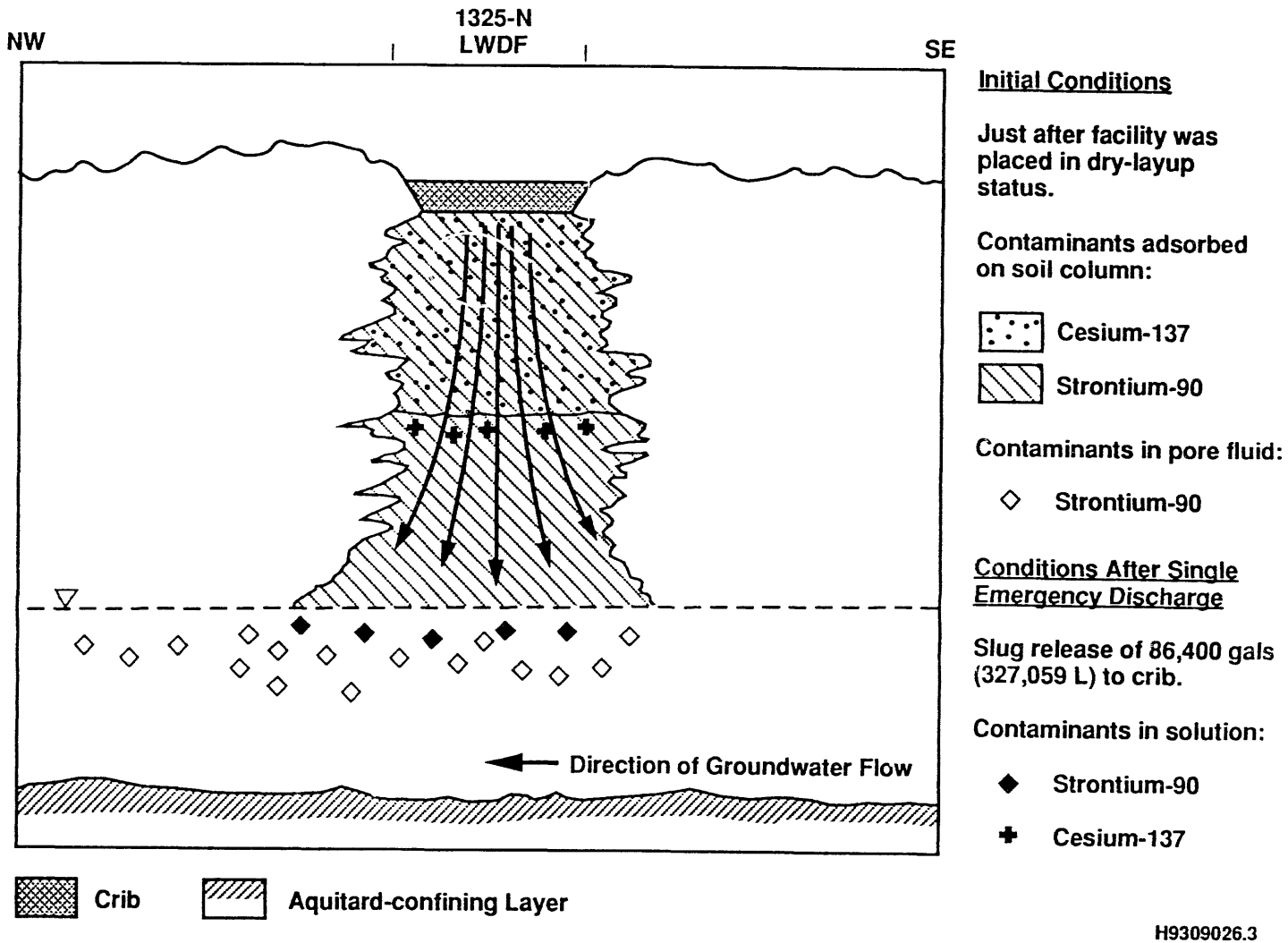
(5) The displaced pore volume and associated strontium-90 and cesium-137 enters the unconfined aquifer, undergoing dilution and dispersion as it slowly migrates toward the river.

(6) The strontium-90 migrates more rapidly than the cesium-137 because of interaction with water softener regenerant chemicals from an adjacent past-practice disposal facility.

The contaminant transport and release scenario summarized above is evaluated in the following section using previous transport model calculations, results of laboratory sorption-desorption studies, and calculation of theoretical pore fluid radionuclide concentrations in the soil column beneath the crib. This model of contaminant migration through the vadose zone is considered conservative because:

- A piston-like displacement of pore fluid is assumed
- The water table has dropped 6 m (20 ft) since breakthrough of the strontium-90 became apparent.

Figure 39. Illustrated Conceptual Model for the 1325-N LWDF.



5.0 IMPACT ASSESSMENT

As required by the methodology document (Tyler 1991), both hydraulic and contaminant impacts are considered for each groundwater impact assessment. Accordingly, hydraulic factors relevant to the current status of the disposal facility are discussed first followed by contaminant transport analysis.

5.1 HYDRAULIC CONSIDERATIONS

The primary hydraulic considerations for evaluation of the conceptual model include discharge volume, moisture status of the soil column, and possible effects of multiple batch releases as discussed in the following sections.

5.1.1 Slug Discharge Volume

If there is an emergency, discharge of associated water is limited by Tri-Party Agreement Milestone M-17-15A. Under these restrictions, the discharge cannot exceed 327,059 L (86,400 gal) for a given calendar month (see Section 3.3).

5.1.2 Moisture Status of Soil Column

The 1325-N LWDF Crib was previously wetted (saturated flow). Wastewater has also overflowed the crib and into the first section of the trench. While the prior "wetting" of the soil column reduces the retention capacity for subsequent batch releases, current conditions may have improved this situation. The facility has not been used since April 1991, and before that time, received greatly reduced amounts of effluent. For the last 27 months, the soil column beneath the crib has been "drying out" or dewatering because of drainage and evaporation. In addition, the dramatic drop in the water table under the 100-N Area (over 6 m [20 ft]) and dissipation of the mound under 1325-N since 1990, increases the thickness of the unsaturated zone beneath the crib. This provides an additional buffer zone between the current water table elevation and the contaminated sediments above the water table.

5.1.3 Response to Multiple Releases

The infiltration area of the 1325-N Crib is 5,574 m^2 (60,000 ft^2); discharging 327,059 L (86,400 gal) to the crib over a 30-day period would only serve to rewet but not saturate the soil column. If the discharges resume on a consistent basis or more than one "emergency" occurs (multiple batch discharges), then transport of pore fluid and associated contaminants down to the water table could occur. (The potential magnitude of the latter effect is estimated in Section 5.2.2.)

5.2 CONTAMINANT IMPACTS

Evaluation of potential impacts of slug releases on groundwater quality in the vicinity of the crib requires:

- Prediction of the status of the radionuclide inventory on the soil column
- Calculation of the equilibrium pore fluid radionuclide concentrations.

A one-dimensional model is used to assess the depth of migration of contaminants in the soil column as described in Section 5.2.1. Pore fluid concentrations are estimated in Section 5.2.2.

5.2.1 One-Dimensional Flow and Transport Analysis (Analytical Methods)

The same one-dimensional analytical method described in the *Liquid Effluent Study Final Project Report* (WHC 1990a) was employed to estimate the rate of moisture and contaminant movement through the soil column beneath the crib during the operational period (1983-1988). The method considers only flow in the vertical direction and does not allow for lateral spreading. Thus it is expected to provide migration rates that are faster than those that occur under actual conditions.

The method is based on steady-state flow conditions in the unsaturated zone and assumes a unit hydraulic gradient. The basic equation for any layer of sediments is

$$t = L \times \theta/q \qquad (1)$$

where:

t = time of travel through layer, seconds
L = thickness of layer, centimeters
θ = moisture content of sediment, related to hydraulic conductivity
q = Darcy velocity or moisture flux in layer, centimeters/seconds.

The total travel time, T, is determined as the summation of the travel times for each of the "i" layers:

$$T = \sum_{i=1}^{n} L_i \times \theta_i/q_i \qquad (2)$$

where n is the number of sediment layers. For transport calculation purposes, the soil column beneath the crib is treated as a single layer of coarse sand and gravel with a depth to water of approximately 20 m (66 ft) (Figure 40).

The relationship between hydraulic conductivity, K, and moisture content, θ, is described graphically in Figure 41. These curves were derived empirically from laboratory tests on over 20 different Hanford Site sediment types, and were used to establish 5 major sediment types, as noted in Figure 41.

The one-dimensional flow analysis embodied in equation 2 was carried out on a Symphony[1] spreadsheet. The total travel time, T, obtained with equation 2 is divided into the vadose zone thickness to provide an estimate of the rate of moisture migration from the disposal facility to the groundwater.

To obtain an estimate of the rate of contaminant migration, the retardation factor, R_f, for each of the contaminants identified was estimated from the following approximation for Hanford Site soils:

$$R_f = 1 + 5K_d \tag{3}$$

The K_d values were selected from Ames and Serne (1991). When the rate of moisture migration is divided by the R_f for the contaminant of interest, the result is an estimate of the contaminant migration rate. These computations were also carried out using the Symphony spreadsheet method.

The effluent discharge rate, as described previously, is entered as liters per month in the spreadsheet computational method. Effluent volumes through 1987 listed in WHC-EP-0287, Vol. 3, were updated to include 1988 and 1989 (WHC 1989b) for the *Liquid Effluent Study Final Project Report* (WHC 1990a; the same average infiltration rate was also assumed for the time period subsequent to 1989). The total volume (liters) was divided by the corresponding operating period (months) to establish an average rate of inflow (L/month). This effluent discharge rate was divided by the crib area to obtain an estimate of the average infiltration rate.

More details and an illustrative example for application of the overall computational approach are provided in WHC (1990a).

5.2.2 Results of Analytical Solution for Operational Period

(The following discussion summarizes the results from the *Liquid Effluent Study Final Project Report* [WHC 1990a]).

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[1]Symphony is a registered trademark of the Lotus Development Corporation.

Figure 40. Lithology of Well 199-N-70 Near the 1325-N Crib.

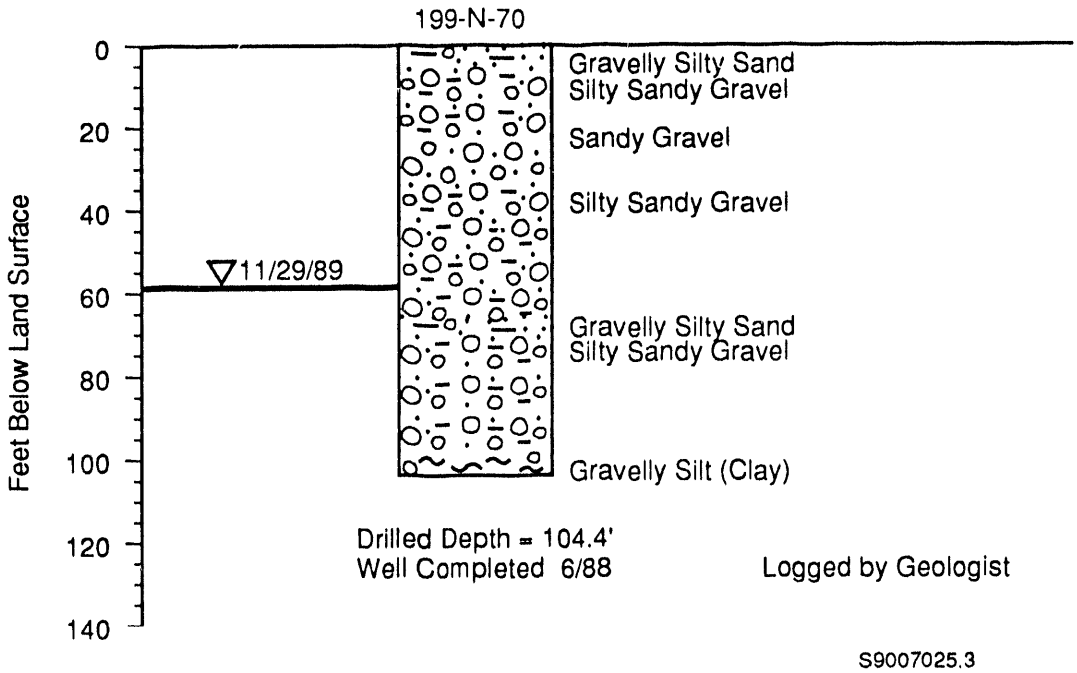
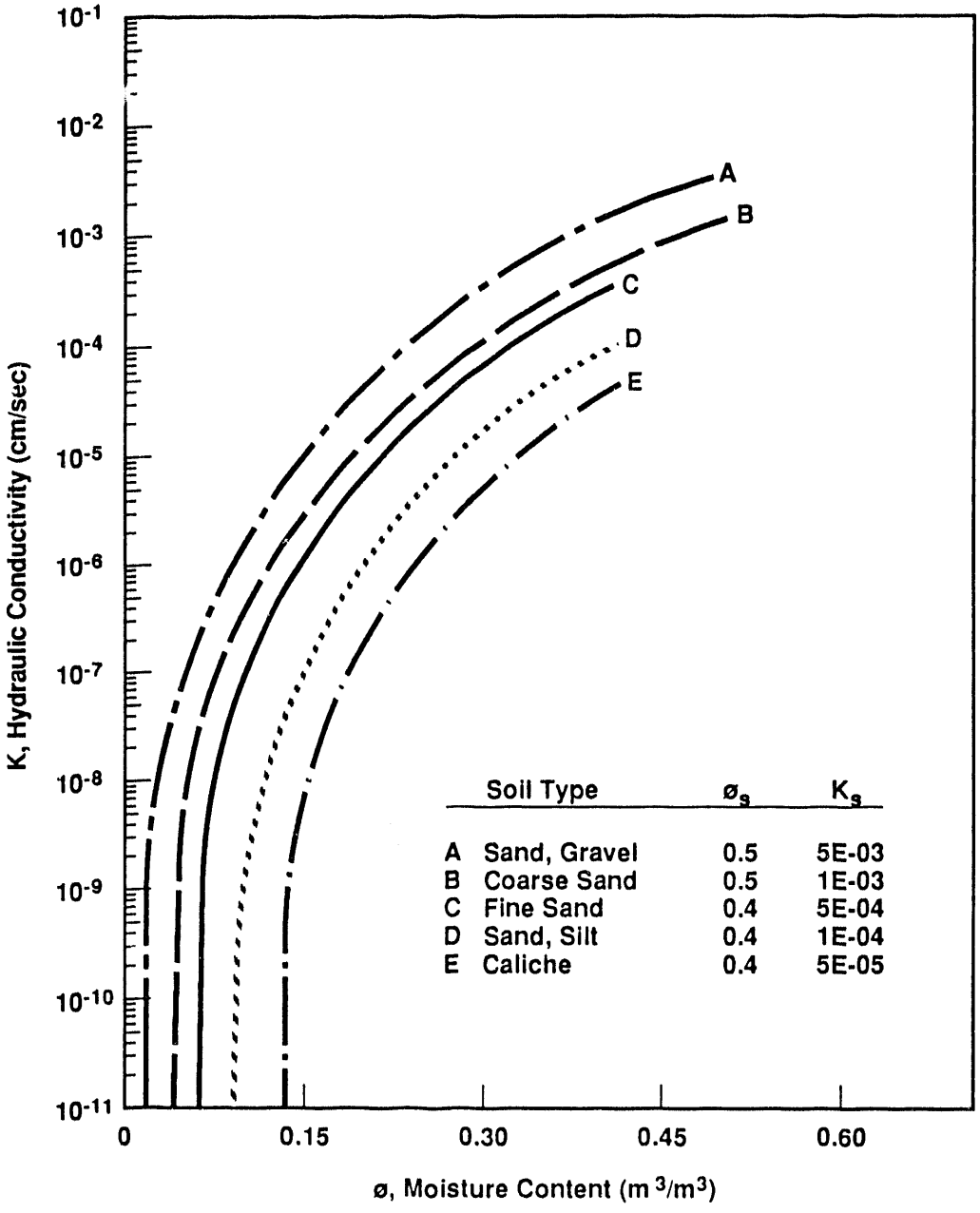


Figure 41. Hydraulic Conductivity Versus Moisture Content (WHC 1990a).



H9309026.5

Based on general effluent characteristics and corresponding sorption parameters (Section 5.2.1) for the key constituents identified in Table 12, the calculated migration rates in the vadose zone (Table 13), listed most mobile to least mobile as follows:

- 153 cm/day (60 in./day) for tritium, antimony-124, antimony-125, and iodine-131
- 2 cm/day (0.8 in./day) for ruthenium-103 and ruthenium-106
- 1 cm/day (0.4 in./day) for plutonium-238, plutonium-239, zinc-65, strontium-89, strontium-90, zirconium-95, niobium-95, molybdenum-99, and barium-140
- 0.6 cm/day (0.2 in./day) for iron, phosphorus-32, manganese-54, cobalt-58, iron-59, cobalt-60, cesium-134, and cesium-137
- 0.2 cm/day (0.08 in./day) for lanthanum-140, cerium-141, and cerium-144.

The most significant constituents based on inventory, half-life, and relative hazard potential are strontium-90 and cesium-137. Based on the migration rates for these constituents (see Table 13) and above, breakthrough of strontium-90 should have occurred approximately 2 years before shutdown of N Reactor while cesium-137 (travel time of 7 years) should still be retained in the soil column. The presence of strontium-90 and the absence of cesium-137 in groundwater in the immediate vicinity of the crib (Section 4.3) are consistent with the corresponding migration rate estimates noted above.

5.2.3 Soil Column Pore Fluid Strontium-90 Concentrations

5.2.3.1 Single Emergency Slug Release. As suggested in Section 4.5, displacement of residual water in the soil pore spaces that has been in equilibrium with adsorbed strontium-90 is one potential pathway for groundwater contamination from occasional emergency water releases to the crib. To estimate this concentration we make use of the definition of K_d:

$$K_d = \frac{C_s,\ \text{pCi/g}}{C_w,\ \text{pCi/mL}} \qquad (4)$$

where:

C_s = the equilibrium concentration of strontium-90 in the solid phase
C_w = the concentration in the aqueous phase in a sediment-water mixture from batch laboratory sorption tests.

Table 12. Effluent Stream Sampling Data (WHC 1990a). (2 sheets)

Key constituents	Detection limit[a]	Detection/ analyses	Sample concentration (90% CI)	Detection limit[a]	Detection/ analyses	Sample concentration (90% CI)
03/88 to 08/88, During cold standby				10/89 to 03/90, During cold standby		
Iron	30	4/4	420	30	4/4	296[b]
1976 to 1988, During reactor operation[c]				10/89 to 03/90, During cold standby		
Tritium	1,000	206/NA	110,000	100	3/3	198,000
Manganese-54	10	231/NA	64,000	NA	3/3	604[b]
Iron-59	10	86/NA	22,000	NA	ND	ND
Cobalt-60	10	236/NA	41,000	1	3/3	700
Zinc-65	100	12/NA	2,500	NA	ND	ND
Strontium-89	100	121/NA	74,000	NA	ND	ND
Strontium-90	200	205/NA	33,000	0.1	2/2	176,000
Ruthenium-106	100	9/NA	31,000	50	1/2	1,720
Iodine-131	100	91/NA	160,000	NA	ND	ND
Cesium-134	10	96/NA	1,500	NA	2/2	167
Cesium-137	10	236/NA	42,000	1	3/3	47,000
Lanthanum-140	10	102/NA	910,000	0.001	ND	ND
Plutonium-239	0.1	206/NA	420	NA	3/3	66
Americium-241	20	ND	ND	NA	3/3	70
Phosphorus-32	NA	136/NA	4,500	NA	ND	ND
Cobalt-58	NA	16/NA	3,000	NA	ND	ND
Zirconium-95	NA	79/NA	11,000	NA	ND	ND
Niobium-95	NA	115/NA	23,000	NA	ND	ND
1976 to 1988, During reactor operation[c]				10/89 to 03/90, During cold standby		
Molybdenum-99	NA	73/NA	400,000	NA	ND	ND
Ruthenium-103	NA	100/NA	39,000	NA	ND	ND
Antimony-124	NA	4/NA	1,600	NA	ND	ND
Antimony-125	NA	8/NA	5,000	NA	ND	ND
Barium-140	NA	99/NA	48,000	NA	ND	ND
Cerium-141	NA	72/NA	15,000	NA	ND	ND

Table 12. Effluent Stream Sampling Data (WHC 1990a). (2 sheets)

Key constituents	Detection limit[a]	Detection/ analyses	Sample concentration (90% CI)	Detection limit[a]	Detection/ analyses	Sample concentration (90% CI)
Cerium-144	NA	106/NA	39,000	NA	ND	ND
Plutonium-238	NA	201/NA	64	NA	3/3	11
Lead-210	NA	ND	ND	NA	1/3	3

[a]Units: chemical = ppb
radionuclides = pCi/L
[b]Concentrations below Group A study guidelines; values given for purposes of comparison.
[c]Disposal of effluent to the 1325-N Crib began in 1983; thus, this data set includes some samples taken before 1983.
CI = confidence interval.
NA = not available.
ND = not detected.

Table 13. Initial Analytical Solution Results for the 1325-N LWDF (WHC 1990a).

Disposal facility	Rate (L/month)	Area (m^2)	f (infiltration rate cm/s)	Number of layers	Thickness (m)	Soil type
1325-N LWDF	8.01 E+07	5,574	5.47 E-04	1	15	A

θ_s	K_s	θ	Moisture State	q (cm/s)	t (s)	T (d)	Estimated moisture migration (cm/d)
0.5	5.0 E-03	0.31	Unsaturated	5.47 E-04	8.50 E+05	10	152.5

Constituent	R_f (retardation factor)	Estimated contaminant migration (cm/day)	Contaminant transport to water table (year)
Iron	250	0.6	7
Plutonium-238	125	1	3
Plutonium-239	125	1	3
Tritium	1	153	0.03
Phosphorus-32	250	0.6	7
Manganese-54	250	0.6	7
Cobalt-58	250	0.6	7
Iron-59	250	0.6	7
Cobalt-60	250	0.6	7
Zinc-65	150	1	4
Strontium-89	125	1	3
Strontium-90	125	1	3
Zirconium-95	200	1	5
Niobium-95	200	1	5
Molybdenum-99	200	1	5
Ruthenium-103	100	2	3
Ruthenium-106	100	2	3
Antimony-124	1	153	0.03
Antimony-125	1	153	0.03
Iodine-131	1	153	0.03
Cesium-134	250	0.6	7
Cesium-135	250	0.6	7
Lanthanum-140	1000	0.2	27
Barium-140	125	1	3
Cerium-141	1000	0.2	27
Cerium-144	1000	0.2	27

LWDF = liquid waste disposal facility.

Assuming equilibrium conditions and that laboratory sorption or K_d data are representative of soil column conditions, pore fluid strontium-90 concentration (pCi/mL) in the wetted portion of the soil column containing the strontium-90 inventory can be estimated be rearranging equation 4 as follows:

$$C_s,\ pCi/mL = \frac{C_w,\ pCi/g}{K_d} \tag{5}$$

where:

C_s = the adsorbed strontium-90 inventory retained on the sediments in the vadose zone beneath the crib
C_w = the pore fluid concentration of strontium-90 in equilibrium with the adsorbed strontium-90 on the sediments.

Ames and Serne (1991) indicate a conservative estimate of 25 for a strontium-90 K_d in Hanford Site soils (effluent with low salt content, slightly basic, no complexants, no organics). This is the same value used in Table 13 to estimate the R_f. Next, an estimate of the strontium-90 concentration in sediments beneath the crib is needed to solve equation 5 for pore fluid concentration. To make this estimate, the following assumptions are used:

- Strontium-90 inventory is uniformly distributed within a volume of sediment defined by the area of the crib
- Depth to the original water table was 15 m (50 ft)
- Porosity of 0.3 and a sediment density of 2 g/mL
- Crib contained a total inventory of 60 Ci.

This implies that all of the strontium-90 added to the crib had been retained and that it was just at the point of entering the water table when discharges were curtailed. Combining these terms the estimated soil (adsorbed) strontium-90 concentration is:

$$C_s,\ pCi/g = \frac{60\ Ci \times 10^{12}\ pCi/Ci}{0.7 \times 15\ m \times 5574\ m^2 \times 1000\ L/m^3 \times 2\ g/mL \times 1000\ mL/L} = 512\ Ci/g$$

A factor of 0.7 in the above calculation is used to account for the fact that the volume of solids in a unit volume of soil is 70% total or bulk volume.

Substituting in equation 5 yields a pore fluid strontium-90 concentration of:

$$C_w,\ pCi/mL = \frac{512\ pCi/g}{25} = 20\ pCi/mL \text{ or } 20{,}000\ pCi/L$$

Assuming that a slug release of volume (V) of 327,059 L (86,400 gal) displaces an equal volume of residual fluid from the vadose zone beneath the crib, the Ci quantity of strontium-90 added to the aquifer from this single event is:

$$\text{Quantity} = V \times C_w = (327{,}059\ \text{L} \times 20{,}000\ \text{pCi/L})/(1 \times 10^{12}\ \text{pCi/Ci}) = 0.0065\ \text{Ci}$$

This estimate suggests that on a relative curie quantity basis (i.e., 0.01% of the inventory and less than 1% of the annual strontium-90 previously entering the river), the potential input of additional strontium-90 to the aquifer is not very significant. However, concentrations in groundwater in the vicinity of the crib, even after significant dilution, would be well in excess of the WWQS for this constituent (8 pCi/L). By comparison, the highest existing groundwater concentration adjacent to the crib today is 343 pCi/L in well N-29. Thus, while residual strontium-90 concentrations in groundwater near the crib already exceed the WWQS because of past-practice disposal operations, the incremental addition caused by a hypothetical slug release of noncontaminated water in the future could result in much higher concentrations but in a much smaller area than occurred during operations (see Figure 37, Section 4.3).

5.2.3.2 Infiltration Because of Precipitation. It is also instructive to consider the potential effect of net drainage from precipitation. Assuming a net infiltration of 5 cm/yr (2 in./yr) distributed over an area the size of the crib, the corresponding volume is:

$$\text{Volume} = 0.05\ \text{m} \times 5570\ \text{m}^2 \times 1000\ \text{L/m}^3 = 278{,}500\ \text{L (73,572 gal)}.$$

This is nearly equivalent to the volume resulting from a single slug release as calculated above. The actual volume of "natural" drainage passing through the contaminated sediments is difficult to determine because the covered area would act as a shield. Drainage from the perimeter of the crib (area from which vegetation is removed) would intersect an unknown portion of the contaminated sediments. Also, the course sands and gravels characteristic of the backfill area around the crib may actually result in infiltration rates approaching the average precipitation rate (about 16 cm/yr [6 in./yr]).

Regardless of the uncertainties as noted above, natural infiltration of moisture through some fraction of contaminated sediments beneath the crib probably occurs. This represents a continuing source of groundwater contamination that on an annual basis may exceed the amount resulting from a random discharge of spent emergency water.

5.2.4 Mitigating Factors and Alternative Assumptions

One mitigating factor for the impact of emergency water disposal, as previously discussed, is that some restoration of water retention capacity of the soil column has probably occurred due to evaporation because intensive use of the crib ceased 2 to 3 years ago. Thus the driving force for the slug release scenario may be less than that assumed for the calculation.

In addition to the above, a less conservative or more realistic K_d value would lower the estimated pore fluid concentration. For example, laboratory sorption studies using sediments from an upgradient location showed that the strontium-90 desorption K_d is a function of the Ca^{2+} and Mg^{2+} concentration (moles/L) described by the following regression equation (see Appendix A):

$$\log K_d = -0.89 \log [Ca + Mg] - 1.06 \qquad (6)$$

Assuming the hypothetical slug release of emergency water has a chemical composition similar to river water (Mg = 6 ppm and Ca = 23 ppm), the K_d for strontium-90 in 100-N Area sediments would be:

$$\log K_d = -0.89 \log[(0.25 + 0.6)* 10^{-3}] - 1.06$$

$$K_d = 190$$

Pore fluid strontium-90 concentrations using the higher or less conservative K_d from equation 6 in equation 5 would yield an equilibrium concentration of:

$$(512 \text{ pCi/g})/190 \times 1{,}000 \text{ mL/L} = 2{,}700 \text{ pCi/L}$$

Thus, depending on the assumptions used and the release scenario and variables as discussed in previous sections, the pore fluid concentrations could range from about 2,000 to 20,000 pCi/L. Even at the lower end of this range, the concentrations would be significantly greater that the WWQS of 8 pCi\L.

In view of the potential magnitude of the increase in local groundwater strontium-90 concentrations from a slug release, the decision to discharge any additional water to this crib should be carefully considered. In addition, some interim action may be appropriate to minimize natural recharge around the cribs. For example, interim restoration of the natural vegetation (Routson and Johnson 1990) to enhance evapotranspiration, may help to reduce the potential for continuing drainage of strontium-90 into the aquifer from this crib.

5.3 EVALUATION OF MONITORING NETWORK ADEQUACY

Three RCRA facilities are located in the 100-N Area: the 1301-N LWDF, 1325-N LWDF, and the 1324-N/NA Facilities (see Figure 24). Interim-status groundwater monitoring began at all three facilities in December 1987 (EPA 1989). The facilities are located close together and have interrelated effects on the groundwater hydrology and chemistry of the 100-N Area (DOE-RL 1992b).

The 1301-N LWDF and 1324-N/NA Facilities were monitored under a groundwater quality assessment program during 1991. The 1325-N LWDF was in a detection program in 1991. The 1301-N and 1324-N/NA Facilities went into assessment in the spring of 1989, because of elevated specific conductance in

some downgradient wells. The groundwater quality assessment program yielded no evidence that 1301-N or 1324-N/NA have contributed hazardous constituents to the groundwater (Hartman 1990). A new groundwater monitoring plan for the three 100-N Area RCRA sites was issued in 1991 (Hartman 1991). No data were available from the 1991 monitoring when the annual report was written (DOE-RL 1992b).

5.3.1 Groundwater Monitoring Well Placement

During 1992, the groundwater in the 100-N Area was monitored quarterly under the RCRA and CERCLA programs. The 1301-N and 1324-N/NA Facilities were monitored under an assessment program and the 1325-N Facility remained under a detection program. Some new wells were added to the monitoring network to replace ones that had gone dry because of the continued drop in water table in the 100-N Area. Tables 14, 15, and 16 list the monitoring wells used for the RCRA and CERCLA programs for the 100-N Area facilities. The tables give information on the following items:

- Age of the well
- Where it is screened
- Its position with the network (e.g., upgradient)
- Sampling frequency
- If water level measurements are also taken
- What standards the wells are constructed to
- If the wells are used by programs other than RCRA (e.g., CERCLA operable unit numbers).

The groundwater monitoring well network for the entire 100-N Area and each of its component RCRA facilities (1301-N and 1325-N LWDFs, 1324-N/NA Facilities) is adequate.

5.3.2 Reporting of Monitoring Data

All hydrochemical monitoring data are reported in the HEIS database, which is publicly accessible. Monitoring results for the wells in the 100-N Area, for the period January 1990 through August 1993, were summarized and used to evaluate the groundwater quality and chemistry in Section 4.3 in the previous section.

Table 14. Wells Used to Monitor Groundwater Chemistry and Water Levels for the 1301-N Liquid Waste Disposal Facility (DOE-RL 1993).

Well	Aquifer	Sampling frequency	Water levels	Well standards	Other networks
199-N-2[64]	Top of unconfined	Q	Y	PRE	--
199-N-3[64]	Top of unconfined	Q	Y	PRE	100-NR-2
199-N-14[69]	Top of unconfined	Q	Y	PRE	--
199-N-17[81]	Top of unconfined	Q	Y	PRE	1324-N/NA ASMT; 100-NR-2
199-N-21[81]	Top of unconfined	Q	Y	PRE	1324-N/NA ASMT; 100-NR-2
199-N-47[84]	Top of unconfined	Q	---	PRE	1324-N/NA ASMT
199-N-54[87]	Top of unconfined	Q	Y	RCRA	1324-N/NA ASMT; 100-NR-2
199-N-55[87]	Top of unconfined	Q	Y	RCRA	1324-N/NA ASMT
199-N-56[87]	Top of unconfined	Q	Y	RCRA	1324-N/NA ASMT
199-N-57[87] U	Top of unconfined	Q	Y	RCRA	1324-N/NA ASMT
199-N-66[87] U	Top of unconfined	Q	Y	RCRA	100-NR-2
199-N-67[88]	Top of unconfined	Q	Y	RCRA	100-NR-2
199-N-69[88]	Bottom unconfined	Q	Y	RCRA	--
199-N-75[92]	Top of unconfined	Q[a]	Y(93)	RCRA	100-NR-2
199-N-76[92]	Top of unconfined	Q[a]	Y(93)	RCRA	100-NR-2

[a]Quarterly sampling began the last quarter of 1992.

Note: Superscript following well number denotes the year of installation.
Y = used to monitor water levels (year in parentheses means monitored beginning in that year).
PRE = well was not constructed to RCRA standards.
Q = frequency on a quarterly basis.
RCRA = well is in compliance with Resource Conservation and Recovery Act of 1976 (RCRA standards).
U = upgradient well in sampling network.
ASMT = assessment.

Table 15. Wells Used to Monitor Groundwater Chemistry and Water Levels for the 1324-N/NA Liquid Waste Disposal Facility (DOE-RL 1993).

Well	Aquifer	Sampling Frequency	Water Levels	Well Stand.	Other Networks
199-N-17[81]	Top of unconfined	Q	Y	PRE	1301-N ASMT; 100-NR-2
199-N-21[81]	Top of unconfined	Q	Y	PRE	1301-N ASMT; 100-NR-2
199-N-47[84]	Top of unconfined	Q	---	PRE	1301-N ASMT
199-N-54[87]	Top of unconfined	Q	Y	RCRA	1301-N ASMT; 100-NR-2
199-N-55[87]	Top of unconfined	Q	Y	RCRA	1301-N ASMT
199-N-56[87]	Top of unconfined	Q	Y	RCRA	1301-N ASMT
199-N-59[87]	Top of unconfined	Q	Y	RCRA	--
199-N-71[91] U	Top of unconfined	Q	Y	RCRA	100-NR-2
199-N-72[91]	Top of unconfined	Q	Y	RCRA	--
199-N-73[91]	Top of unconfined	Q	Y	RCRA	100-NR-2
199-N-77[92]	Bottom unconfined	Q[a]	Y(93)	RCRA	100-NR-2
699-81-58[62] U	Top of unconfined	Q	Y	PRE	1325-N

[a] Quarterly sampling began the last quarter of 1992.
Superscript following well number denotes the year of installation.
ASMT = assessment.
PRE = well was not constructed to RCRA standards.
Q = frequency on a quarterly basis.
RCRA = well is in compliance with *Resource Conservation and Recovery Act of 1976* (RCRA standards).
U = upgradient well in sampling network.
Y = used to monitor water levels (year in parentheses means monitored beginning in that year).

Table 16. Wells Used to Monitor Groundwater Chemistry and Water Levels for the 1325-N Liquid Waste Disposal Facility (DOE-RL 1993).

Well	Aquifer	Sampling Frequency	Water Levels	Well Stand.	Other Networks
199-N-27[83a]	Top of unconfined	SA	Y	PRE	--
199-N-29[83a]	Top of unconfined	SA	Y	PRE	--
199-N-31[83]	Top of unconfined	SA	Y	PRE	--
199-N-32[83]	Top of unconfined	SA	Y	PRE	100-NR-2
199-N-33[83]	Top of unconfined	SA	Y	PRE	--
199-N-41[84]	Top of unconfined	SA	Y	PRE	--
199-N-70[88]	Bottom unconfined	Q	Y	RCRA	100-NR-2
199-N-74[91] U	Top of unconfined	Q	Y	RCRA	100-NR-2
699-81-58[62] U	Top of unconfined	Q	Y	PRE	1324-N/NA ASMT

[a]Although wells N-27 and N-29 are currently downgradient of the 1325-LWDF, they were downgradient when the facility was active. The groundwater chemistry at these wells is still affected by the facility, so they are monitored as downgradient wells.

Note: Quarterly sampling began the last quarter of 1992.

Superscript following well number denotes the year of installation.

ASMT = assessment.
PRE = well was not constructed to RCRA standards.
Q = frequency on a quarterly basis.
SA = frequency on a semi-annual basis.
RCRA = well is in compliance with *Resource Conservation and Recovery Act of 1976* (RCRA) standards.
U = upgradient well in sampling network.
Y = used to monitor water levels (year in parentheses means monitored beginning in that year).

6.0 SUMMARY AND CONCLUSIONS

One-dimensional or unit gradient transport and flow calculations suggest that strontium-90 should have reached the water table during the operational period of use for the 1325-N Crib. The occurrence of strontium-90 in groundwater in the immediate vicinity of the crib is consistent with the predicted breakthrough, supporting the assumptions and methods used for estimating contaminant migration rates through the soil column.

The continuing migration of strontium-90 into the aquifer may be occurring caused by natural recharge. The estimated magnitude of additional strontium-90 input to the aquifer because of single hypothetical slug releases of spent emergency water appears to be about the same order of magnitude as the annual input caused by natural recharge. The localized concentrations because of both types of input exceed the WWQS of 8 pCi/L for strontium-90 and mitigating measures for controlling the natural infiltration around the crib (e.g., restoration of natural vegetation around the crib or installation of an interim infiltration barrier) should be investigated as part of ongoing remediation efforts. Likewise, emergency water disposal should not be discharged to the crib.

Other groundwater constituents with elevated concentrations in the 100-N Area are nitrate, tritium, sulfate, sodium, and chloride. These parameters are being investigated by the RCRA Monitoring Program as part of ongoing investigations within the 100-N Area. Tritium contamination is also being addressed as part of a 100-N Area Expedited Response Action. The groundwater monitoring well network for this site is adequate.

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APPENDIX A

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Letter Report for Westinghouse Hanford Company

ADSORPTION AND DESORPTION OF Sr-90 ON HANFORD SEDIMENT
STATUS REPORT

DRAFT

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R. J. Serne

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the U.S. Department of Energy
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SUMMARY

Introduction

Despite the fact that the flux of radionuclides and water into the 1301-N and 1325-N disposal trenches has diminished significantly since the shutdown of the N Reactor, the observed concentrations of ^{90}Sr in the compliance well N-8t have shown a slight increase in ^{90}Sr concentration from March 1991 to May 1992. One explanation for the observed increase in ^{90}Sr concentrations is that the chemical composition of water flowing through contaminated sediments has changed significantly and is now causing previously adsorbed ^{90}Sr to desorb. Sr adsorption is known to be quite sensitive to competing cations such as Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, and H^{+}. This study was initiated to gain a better understanding of the adsorption/desorption properties of strontium onto/from 100-N Area sediments and how changes in groundwater chemistry can effect these properties.

Available Chemical Data

The available historical chemical data from the 100-N Area, which are useful from a geochemical perspective, are sparse. The lack of historical major ion data, in particular, has hampered a thorough geochemical analysis of the competitive adsorption/desorption reactions that occur between ^{90}Sr and major cations. Some interesting results, however, are available for well N-3. For example, in Figure 1, ^{90}Sr concentrations in Well N-3 are plotted as a function of time along with Ca concentrations. The increase in ^{90}Sr concentration appears to be well correlated with an increase in Ca + Mg concentration. These results are consistent with our hypothesis that ion-exchange of Ca and Mg for previously adsorbed Sr is responsible for the increases in ^{90}Sr observed in well N-8t. Other major ion data from N-Area indicate that high concentrations of ion-exchange regenerant solutions disposed in 1324-N/NA were over 1 Molar in Na concentration. Disposal of these solutions directly into the vadose zone has resulted in greatly elevated concentrations of sodium in the groundwater in the vicinity of the 1324-N/NA disposal site. The elevated sodium concentrations in the groundwater causes desorption of Ca from exchange sites, resulting in elevated Ca concentrations.

Equilibrium Batch Sorption Results

K_d values determined in the batch adsorption and desorption experiments are listed in Table 1. The K_d values listed in the K_d(ads) column were determined after three days of adsorption. Values listed under K_d(des1) were determined after 16 days of desorption. Both K_d(ads) and K_d(des1) values were calculated using a mass balance approach based upon the initial and final concentration values of ^{85}Sr in solution. K_d values under the K_d(des2) column were determined directly by measuring the concentrations both in solution and in the sediment after the 16 day equilibration period. The quality of the K_d data which were determined directly are clearly much more reproducible. For example, the experiments 13 through 17 all have the same initial calcium and magnesium concentrations and we would therefore expect the K_d results to be very similar for all of these experiments. The K_d(des1) results for these

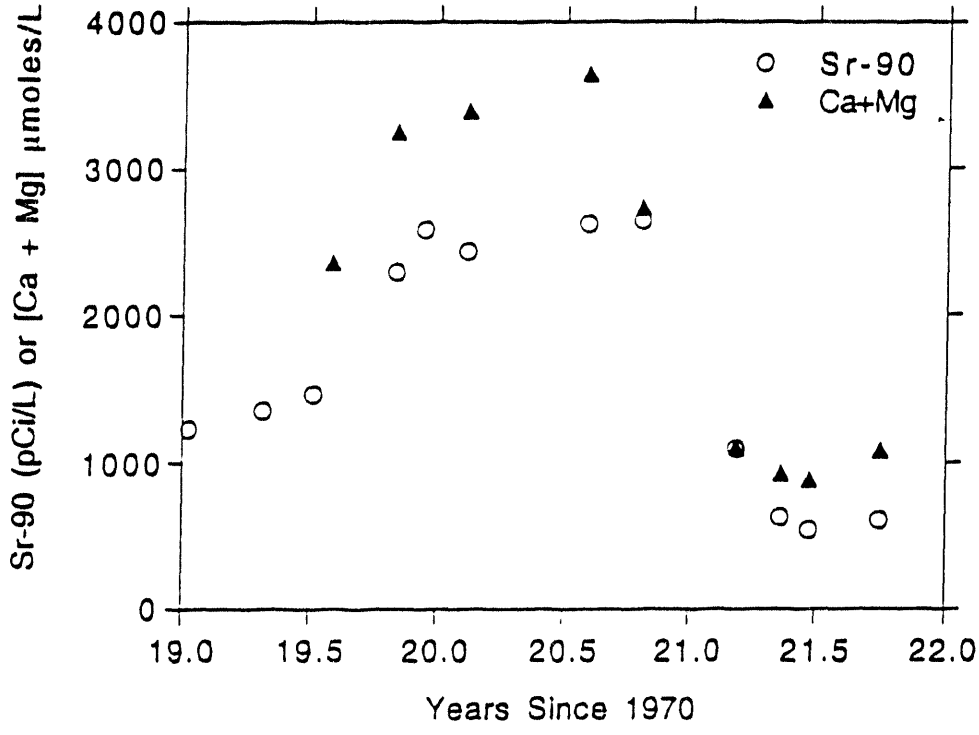

Figure 1. Sr-90 (pCi/L) and the sum of Ca and Mg (µmoles/L) concentrations in well N-3 as function of time.

experiments ranged from -11 to 46. This is a large difference. The K_d(des2) results,however, ranged from 10 to 14. The K_d(des2) results are plotted in Figure 3 as a function of the final solution concentration of Ca plus Mg. We can see that a very good linear correlation exists. These results support our hypothesis that Sr is adsorbed to ion-exchange sites and that Ca and Mg are effective competitors for these sites. These results also suggest that, with our geochemical model, we should be able to predict the future concentrations of ^{90}Sr in groundwater by combining current ^{90}Sr and major ion data from various 100-N Area wells with modeling results of the future movement of the high salt content plume from the 1324 waste disposal pit. Major ion concentrations and pH values measured at equilibrium for these experiments are listed in Table 2.

Table 1. Sr-85 adsorption and desorption K_d values (mL/g).

Exp.#	K_d(ads)	K_d(des1)	K_d(des2)
1	96	111	-
2	18	25	50
3	50	85	102
4	19	40	43
5	105	145	220
6	24	45	62
7	42	82	122
8	21	42	39
9	57	114	157
10	25	50	47
11	2.7	-6.3	12
12	2.7	1.0	12
13	6.8	26	14
14	1.3	-11	11
15	5.5	20	11
16	11	46	13
17	10	42	10
18	10	12	20
19	7	20	17
20	8	20	13
21	10	24	15
22	8	5	24

Table 2. Measured total equilibrium Ca, Mg, Na, NO_3, and SO_4 solution concentrations (millimoles/liter) and pH for and Sr-85 adsorption and desorption experiments.

Exp.#	Ca	Mg	Na	NO_3	SO_4	pH
1	0.182	0.016	0.032	0.000	0.000	7.38
2	0.724	0.016	0.020	1.274	0.000	7.32
3	0.195	0.185	0.005	0.339	0.000	7.45
4	0.773	0.185	0.022	1.597	0.000	7.33
5	0.137	0.012	0.783	0.000	0.323	7.22
6	0.724	0.016	0.783	1.290	0.375	7.24
7	0.160	0.152	0.783	0.339	0.375	7.18
8	0.873	0.177	0.105	1.903	0.312	7.03
9	0.145	0.078	0.391	0.161	0.187	7.34
10	0.724	0.095	0.396	1.419	0.187	7.08
11	3.27	0.012	3.96	6.725	2.020	6.90
12	3.27	1.73	3.96	9.661	1.853	7.01
13	3.29	0.864	0.022	8.015	0.000	7.16
14	3.24	0.864	7.79	7.967	3.727	6.98
15	3.24	0.864	3.92	7.983	1.791	6.94
16	3.29	0.864	3.96	8.145	1.832	7.01
17	3.27	0.864	3.96	8.209	1.832	6.99
18	1.70	0.012	0.017	3.242	0.031	7.26
19	2.00	0.864	3.96	5.467	1.791	7.05
20	1.67	1.67	11.2	6.741	5.517	7.05
21	1.70	1.73	0.017	6.951	0.000	7.24
22	1.70	0.012	11.0	3.113	5.247	7.36

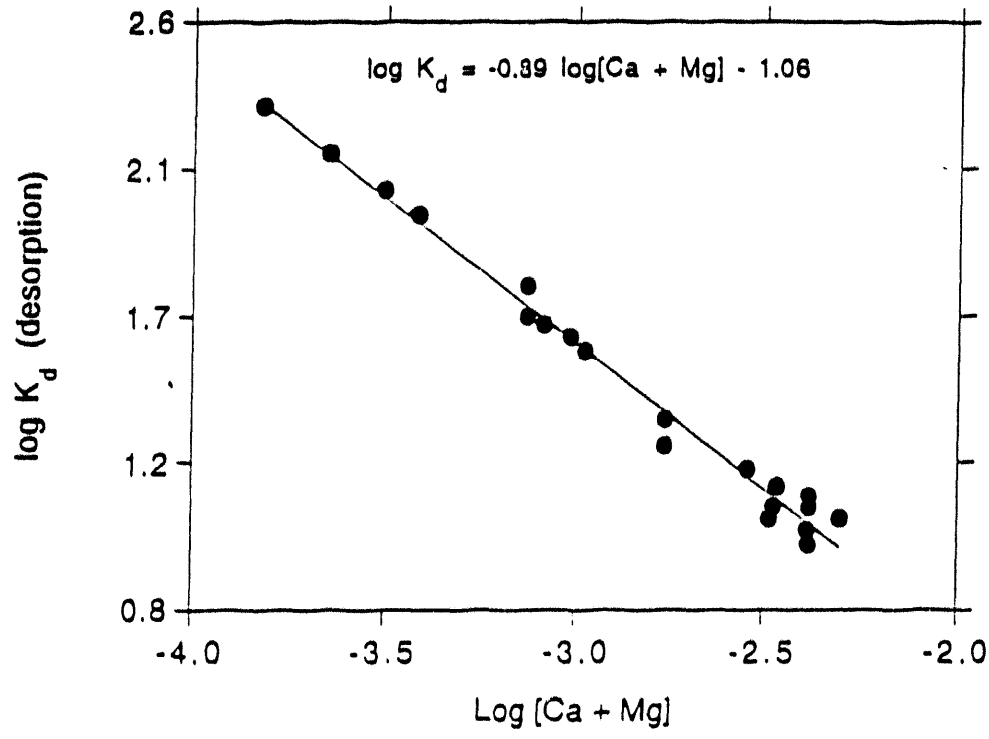

Figure 3. Sr-85 K_d values as a function of the sum of the equilibrium calcium and magnesium concentrations in solution for 100-N Area sediment.

Kinetic Batch Sorption Results

Several attempts have been made to determine reliable adsorption kinetics data which could be used along with the equilibrium K_d results as a check on the column adsorption/desorption modeling results which will be discussed later. In our first attempt to determine the kinetics of strontium adsorption, the adsorption solutions had such a high calcium concentrations that essentially no strontium adsorption occurred. In our second attempt, the initial calcium concentration was very low. However, calcium dissolved from the sediment during the experiment, which caused the equilibrium K_d to change during the course of the experiment. As a result the usefulness of the data was compromised. Results from our last set of kinetics experiments are shown in Figure 2. Although we can see that equilibrium is reached in about two weeks or less, the uncertainty of the data is greater than desired.

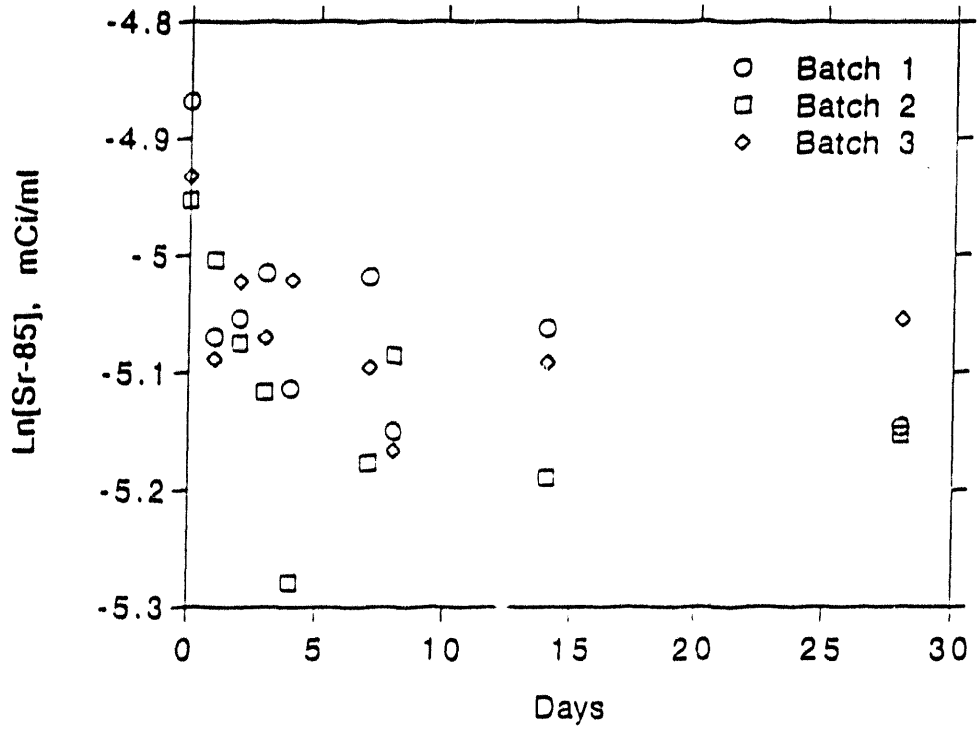

Figure 2. Natural log of the solution concentration of Sr-85 in the adsorption kinetics experiments as a function of time.

Ion-Exchange as the Mechanism of Strontium Adsorption onto Hanford Sediments

Interpretation of the K_d results in terms of an ion-exchange mechanism can be explained as follows. The thermodynamically rigorous mass action equilibrium expression for a binary cation-exchange reaction, such as strontium or cesium adsorbing onto a calcium saturated clay, is

$$aN^{b+} + b(MX) \leftrightarrow a(NX) + bM^{a+} \qquad (1)$$

where a = valence of macro ion
N = trace component (e.g., strontium)
b = valence of trace ion
M = macro component in binary system (e.g., calcium)
X = solid adsorbent (100-N Area sediment)

The equilibrium constant, K, can be expressed as

$$K = ([NX]^a[M^{a+}]^b)/([MX]^b[N^{b+}]^a) \qquad (2)$$

where brackets, [], indicate thermodynamic activities. If one assumes that the exchange capacity, C, of the solid adsorbent is constant (equivalents per unit weight) and that N is present at low (trace) concentration, then the concentration of the trace constituent adsorbed, (NX), is much smaller than C, and the concentration of macro ions adsorbed on exchange sites, (MX), is approximately equal to C/a [i.e., [MX] ≈ C/a] in terms of moles per unit of weight, because C = (NX)b + (MX)b. Then, by using the standard definition of the distribution coefficient, $K_d = (NX)/(N^{b+})$, where (N^{b+}) is the solution concentration of the trace constituent at equilibrium with the solid, and by substituting the relationship that activity is equal to the concentration times an activity coefficient, i.e., $[A] = \gamma(A)\bullet(A)$, one can rewrite equation 2 as

$$K = [(K_d)^a(M^{a+})^b/(C/a)^b]\bullet\Gamma \qquad (3)$$

where Γ is the ratio of the activity coefficients:

$$\Gamma = [\gamma^a(NX)\gamma^b(M^{a+})/[\gamma^b(MX)\gamma^a(N^{b+})] \qquad (4)$$

For ideal ion exchange of a trace constituent where the exchange capacity, C is constant, the ratio of activity coefficients for the adsorbed ions, $\gamma^a(NX)/\gamma^b(MX)$, is constant. Further, for low ionic strength solutions the ratio $\gamma^b(M^{a+})/\gamma^a(N^{b+})$ is also constant, and thus Γ becomes constant. Using these conditions/assumptions and a loganthmic transform of equation 3, the dependence of K_d (the trace constituent distribution coefficient) on the macro ion concentration, (M), reduces to -b/a, the ratio of tracer ion charge to the macro ion charge. For exchange of Sr^{2+} for Ca^{2+}, -b/a is -1.0.

A linear regression of the data in Figure 1 gives the following equation

$$\log K_d = -0.89\log[Ca + Mg] - 1.058 \qquad (5)$$

The slope -0.89 is close to the theoretical value of -1.0, which suggests that ion-exchange is the dominant adsorption mechanism for strontium; however, it is different from -1.0 by an amount which suggests that another mechanism may also be operative.

Column Experiment Results

In order to verify the validity of the adsorption data discussed in the previous sections of this paper, three column experiments were conducted. Each experiment was run at a flow rate of approximately 5 pore volumes per day. The first two column experiments (H1 and H2) were conducted using a solution concentration of 0.00312 M $Ca(NO_3)_2$, 0.000823 M $Mg(NO_3)_2$ and 0.00156 M Na_2SO_4. The third column (L3) was conducted using solution concentrations of 0.000624 M $Ca(NO_3)_2$, 0.000165 M $Mg(NO_3)_2$ and 0.000312 M Na_2SO_4. Each experiment was initiated by conducted by pumping approximately one pore volume of Sr-85 spiked influent

solution. Subsequently, unspiked solution was pumped through the column for the duration of the experiment. The results of these experiments are plotted in Figures 4 to 6, as C/C^o for Sr-85 as a function of pore volume of effluent. Also shown on these figures are model fits to the data using two-site/two-region model(Parker and van Genuchten, 1984). In this model the adsorption term is assumed to consist of two components, one governed by equilibrium adsorption and one by first-order kinetic non-equilibrium adsorption. Modeling parameters are listed in Table 3. R is the retardation factor. D is the dispersion coefficient. β and ω are dimensionless variables from which the first-order rate coefficient, α, is determined.

In the case of columns H1 and H2 the modeling was conducted by fixing the retardation factor to values expected from the equilibrium K_d values determined in the batch adsorption experiments. The adsorption rate coefficients determined in this manner for columns H1 and H2 are very reproducible and are consistent with the results from the batch kinetic experiments.

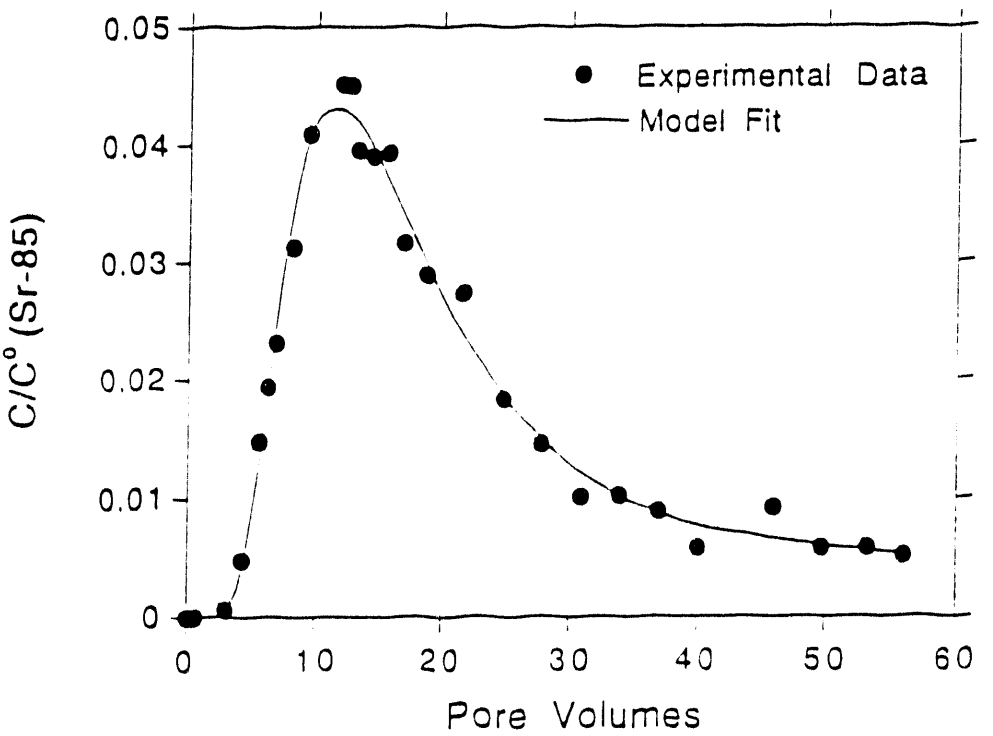

Figure 4. Experimental Sr-85 adsorption/desorption results and model fit for column H1.

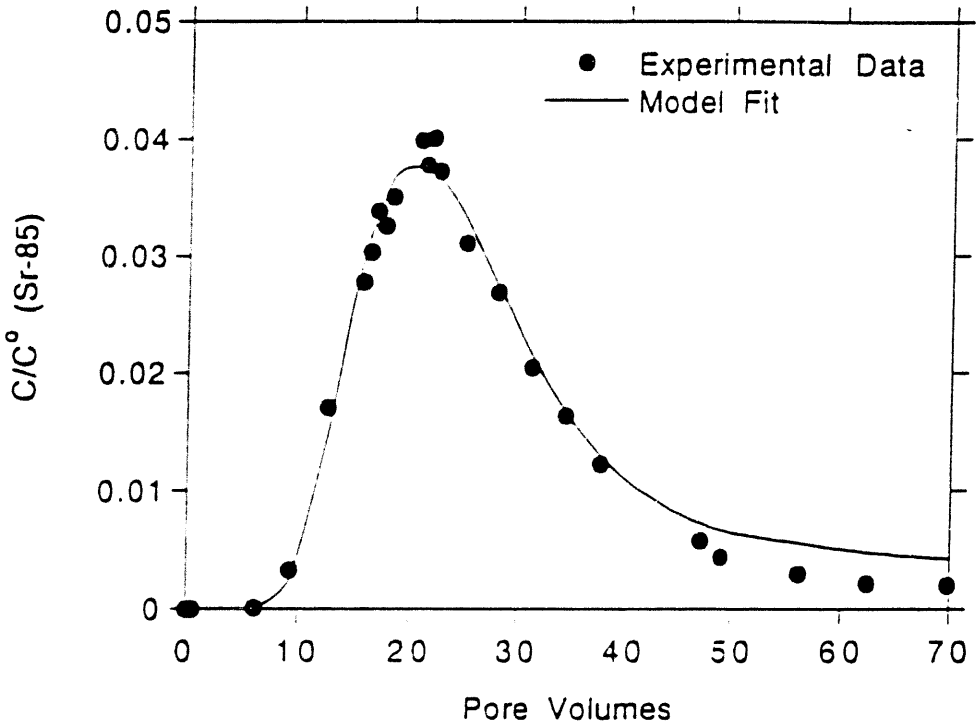

Figure 5. Experimental Sr-85 adsorption/desorption results and model fit for column H2.

In the case of column L1, the expected retardation factor was 292. When this value was fixed, the model did not converge to a reasonable fit of the data. When the model was allowed to fit for the retardation factor, the model converged to a reasonable fit but the fitting parameter results were not reasonable. The retardation factor was determined to be 2376 and α was determined to be 0.005 days^{-1} The retardation factor determined by the model fit was eight times larger than that expected from our batch adsorption experiments. In addition the α determined by the model fit was smaller than that determined by the fits of the H1 and H2 column data by a factor of 20. The reason for the problems in fitting the L1 data have not yet been resolved.

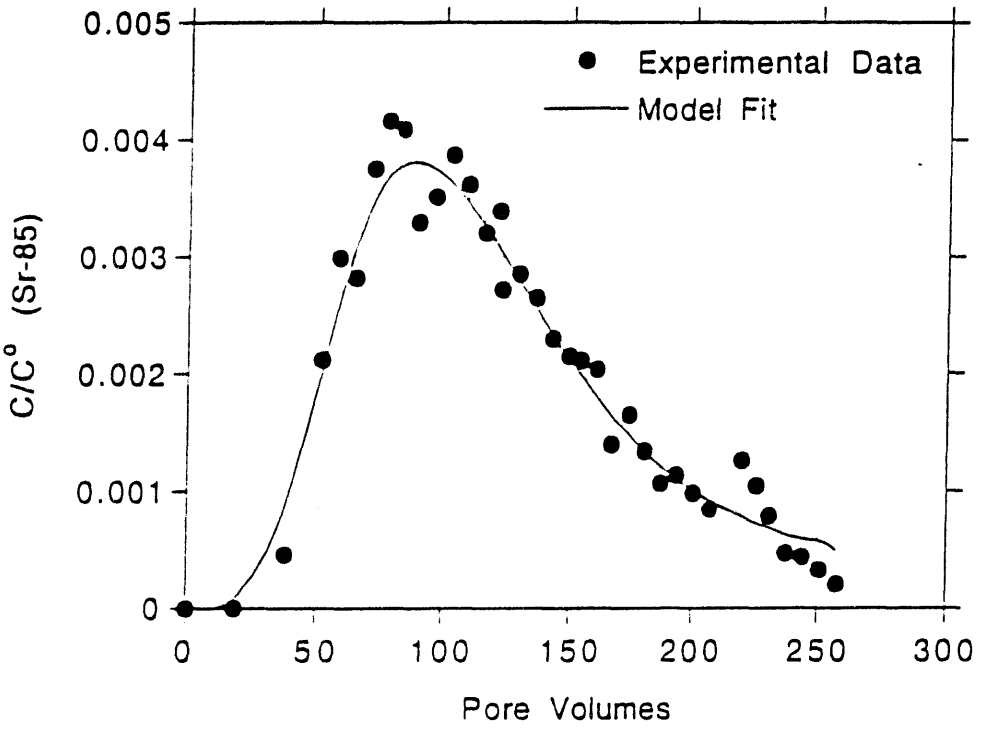

Figure 6. Experimental Sr-85 adsorption/desorption results and model fit for column L1.

Table 3. Modeling parameters determined for column transport experiments.

Column	R	D (cm^2/day)	β	ω	α ($days^{-1}$)
H1	66.4 (held constant)	27.3	0.307	0.706	0.095
H2	67.0 (held constant)	10.4	0.379	0.662	0.100
L1	2376	22.5	0.067	1.56	0.005

References

Parker, J. C. and M. Th. van Genuchten. 1984. Determining Transport Parameters from Laboratory and Field Tracer Experiments. Virginia Agricultural Experiment Station Bulletin 84-3, Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

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