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ESTIMATION OF THE RELEASE AND MIGRATION OF NICKEL
THROUGH SOILS AND GROUNDWATER AT THE HANFORD SITE
218-E-12B BURIAL GROUND

K. Rhoads	L. H. Sawyer
B. N. Bjornstad	J. L. Smoot
R. E. Lewis	J. E. Szecsody
S. S. Teel	M. S. Wigmosta
K. J. Cantrell	S. K. Wurstner
R. J. Serne	

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Pacific Northwest Laboratory
Richland, Washington 99352

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ABSTRACT

This study was performed to evaluate the potential for transport of nickel radioisotopes from the Hanford Site 218-E-12B Burial Ground to the surrounding surface- and groundwater. Burial of large metal components containing nickel-bearing alloys at this location may eventually result in release of elemental nickel and nickel radioisotopes (^{59}Ni and ^{63}Ni) to the subsurface environment, including groundwater aquifers that may be used for domestic and agricultural purposes in the future, and, ultimately, to the Columbia River.

The rate at which nickel could be transported to downgradient locations depends on a complex set of factors, including climate, soil and groundwater chemistry, and the geologic and hydrologic configuration of the subsurface region between the burial ground and a potential receptor location. The geologic structure of the sedimentary formation in this area was investigated using available published information, by observing the walls of the excavated burial trench, and by examining borehole cuttings taken in the region. Physical, hydraulic, and geochemical properties of the sedimentary deposits were determined by laboratory analysis of samples taken from the trench walls and a limited number of samples from borehole cuttings. Release of nickel compounds to the subsurface environment was based on either the corrosion rates of nickel alloys in the components or the solubility limits for the resulting nickel corrosion products in Hanford groundwater. The corrosion rates for nickel alloys were taken from published literature, and the solubility of nickel compounds was determined in laboratory studies.

The groundwater transport analysis was conducted using a one-dimensional screening model with a relatively conservative matrix of parameters obtained from the hydrogeologic and geochemical studies. The predicted peak concentrations of nickel in the unconfined aquifer ranged from 1.9×10^{-16} to $2.0 \text{ pCi/L } ^{59}\text{Ni}$ and 3.1×10^{-4} to $5.1 \times 10^{-2} \text{ mg/L}$ total elemental nickel, depending on assumptions about the number of components buried at the site, the climate, and the receptor location. The estimated transfer of nickel to the Columbia River was less than $1.3 \times 10^7 \text{ pCi/yr}$ for ^{59}Ni and 2.2 kg/yr for

elemental nickel in all cases, resulting in surface-water concentrations that were below 1.3×10^{-7} pCi/L ^{59}Ni or 2.2×10^{-8} mg/L elemental nickel. The times predicted for peak nickel concentrations in the unconfined aquifer and the river ranged from 68,000 to over 4 million years. Because of its short radiological half-life relative to the transport times, ^{63}Ni would not be present at measurable concentrations in the aquifer beneath the burial ground or at downgradient locations.

The maximum dose to an individual using well water from the aquifer to operate a family farm was estimated to be 0.001 mrem/yr or lower, and the 10,000-year collective dose to the downriver population would be less than 5.3×10^{-4} person-rem in all cases. The collective dose from ^{59}Ni entering the Columbia River would be substantially lower than the dose expected to result in a single cancer fatality (2000 person-rem) over the 10,000-year period. For perspective, natural background radiation would result in a dose of approximately 9 billion person-rem to the estimated 3 million people living along the river over a period of 10,000 years.

EXECUTIVE SUMMARY

An assessment was performed to evaluate release and transport of nickel from large metal components containing nickel-bearing alloys at the Hanford Site 218-E-12B Burial Ground. The potential for nickel within the components to enter groundwater under the burial site was investigated by examining available data on the site's geology, geochemistry, and geohydrology to develop a conceptual model for release and transport of nickel from the components. In addition, laboratory studies were performed to provide information needed for the model, but which was not available from existing databases. Estimates of future concentrations of nickel radioisotopes (^{59}Ni and ^{63}Ni) and total elemental nickel in the unconfined aquifer and in the Columbia River were developed based on this information.

The geological strata underlying the burial ground form a sedimentary deposit known as the Hanford formation, a heterogeneous structure that contains layers ranging from fine-grained sand, clay, and silt to gravel and boulders. Beneath the Hanford formation is an extensive, relatively impermeable basalt formation that constitutes the bottom of the unconfined aquifer. Water is assumed to percolate downward through the burial site and the underlying layers of sedimentary deposits until it encounters the unconfined aquifer approximately 61 m (200 ft) below the surface. After the infiltrating water reaches the unconfined aquifer, it is transported by groundwater flow within the aquifer, which eventually discharges to the Columbia River.

The physical and hydraulic properties of soil determine the rate at which rainwater on the surface is transported downward toward the unconfined aquifer, and they also influence the rates of competing processes such as evaporation and transpiration by plants. The characteristics of the Hanford formation beneath the burial site were investigated using a number of existing resources, and by sampling soil from the excavated faces of the burial pit. Strata in the faces of the pit were mapped, and drilling logs from boreholes and wells adjacent to the burial site were used to map soil in the strata between the floor of the pit and the basalt formation. Soil samples collected at the burial pit and a limited number of samples from borehole cuttings were

tested to determine their physical and hydraulic properties, including grain size distribution, moisture content, porosity, permeability, and bulk density. The results of these tests were then used to predict the properties of similar formations in the deeper strata, and ultimately the travel time required for water to reach the unconfined aquifer.

The chemistry of soil and groundwater also play an important role in predicting transport of nickel from the burial site. The release rate of nickel from the metal components depends largely upon the corrosion rates of nickel alloys in the components, the solubility of secondary nickel corrosion products in groundwater, and the total quantity of water percolating through the soil surrounding the components. After dissolution, transport of nickel from the burial ground to the aquifer below would be strongly influenced by the ability of the soil column to adsorb and retain it. The extent to which dissolved nickel could be adsorbed onto soil particles is a relatively complex function of the subsurface water and soil chemistry, and of the properties of the nickel species in solution. For this evaluation, soil samples from the burial site were analyzed to determine their chemical and mineralogical make-up. The chemistry of groundwater in the vicinity was assumed to be consistent with samples taken at onsite monitoring wells.

Release of nickel from the components to the subsurface environment depends on the relative rates of corrosion of the nickel-bearing alloys and dissolution of the resulting corrosion products. The primary nickel-bearing alloys in the components are HY-80, Type 304 stainless steel, and Inconel Alloy 600. For nickel alloys that have very low corrosion rates in the subsurface environment, the corrosion rate would limit nickel release, and dissolution of any corrosion products was assumed to take place more rapidly than their formation. Corrosion rates for the more resistant alloys in the buried components--Type 304 stainless and Inconel 600--were estimated based on a literature search and extrapolated to conditions at the Hanford Site. The corrosion rate for Type 304 stainless steel at Hanford was estimated to be less than $0.02 \text{ mg/dm}^2\text{-yr}$, and the corresponding corrosion rate for Inconel 600 was assumed to be 70% of that for 304 stainless based on their relative corrosion rates in sea water.

For the less corrosion-resistant nickel-bearing alloy, HY-80 steel, release of nickel to the subsurface environment would be controlled by dissolution of the corrosion products, because the corrosion products form more rapidly than they can be transported away by infiltrating groundwater. The solubility of nickel (as nickel hydroxide, the most probable corrosion product) was predicted to be 2 mg/L based on laboratory studies in an aqueous solution that simulated the composition of Hanford groundwater. For the transport modeling, it was assumed that all water leaching through the area occupied by the buried components would dissolve nickel corrosion products up to the saturation limit.

Once dissolved, nickel would also be removed from groundwater by adsorption onto the soil through which the solution passes. Adsorption of nickel onto soil from the burial site was investigated using two methods. In batch adsorption tests, measured quantities of soil were placed into contact with nickel solution of known concentration for a variable length of time, and the distribution coefficient was determined from the relative amounts of nickel remaining in solution and adsorbed to the soil at the end of the contact period. Adsorption was also measured in a dynamic flow-through column test system to more accurately simulate actual conditions at the burial site and to confirm the results of batch tests. Based on the results of batch experiments, the nickel distribution coefficients were used to model a Freundlich adsorption isotherm for incorporation into the transport analysis.

Because a substantial quantity of lead is present in the components, the potential for lead leaching from the components to tie up adsorption sites in the soil and interfere with adsorption of nickel was investigated. This possibility was considered because reduced adsorption of nickel in the soil would result in accelerated transport to downgradient groundwater locations. In order to determine the extent of any interaction between nickel and lead in the Hanford soil and groundwater system, additional batch adsorption experiments for nickel were conducted in the presence of dissolved lead at concentrations near the lead solubility limit. The distribution coefficients for nickel in these experiments were virtually identical to those from experiments in which lead was not present, and it was concluded that lead in

the components would not be likely to affect nickel transport in this geologic system.

In this assessment, transport modeling for nickel utilized data on soil properties, the predicted release rates of nickel from alloys in the buried components, and estimates for adsorption of nickel under the geochemical conditions at the burial site as described previously. The calculations also included source terms consisting of a single buried component or an array of 120 units. Modeling of water flow from the burial site through the vadose zone to the unconfined aquifer included scenarios for three different recharge rates, each of which was coupled to a flow model for the unconfined aquifer appropriate to that case. A recharge rate of 0.1 cm/yr through the vadose zone was used to represent a situation in which the burial site might be protected by an engineered barrier; a recharge rate of 0.5 cm/yr was used to model an unprotected burial site under arid climatic conditions such as those that currently exist; and a recharge rate of 6.0 cm/yr was used to evaluate the potential effect of a more humid climate on nickel transport from an unprotected burial site in the future.

The concentration of nickel in the unconfined aquifer was evaluated at two downgradient locations, 100 m and 5000 m from the burial site, and the quantity of nickel entering the Columbia River annually was also estimated. A distance 100 m downgradient from the burial ground represents the presumed boundary of the disposal site when institutional control is no longer maintained (DOE 1990). The 5000-m distance represents the location of the family farm for an exposure scenario that has been evaluated in previous analyses (DOE 1987; 1989).

The predicted peak concentrations of nickel in the unconfined aquifer for a single component buried at the disposal site ranged from 1.9×10^{-16} to 0.36 pCi/L ^{59}Ni and 3.1×10^{-4} to 0.0092 mg/L total elemental nickel; those for an array of 120 components were between 1.8×10^{-15} and 2.0 pCi/L ^{59}Ni and 0.0017 to 0.051 mg/L elemental nickel, depending on the assumed recharge rate and receptor location. In all cases, the estimated transfer of nickel to the Columbia River was less than 1.3×10^7 pCi/yr ^{59}Ni and 2.2 kg/yr elemental nickel resulting in surface-water concentrations that were below

1.3×10^{-7} pCi/L ^{59}Ni or 2.2×10^{-8} mg/L total elemental nickel. The times required to reach peak nickel concentrations in the unconfined aquifer and the river ranged from 68,000 to over 4 million years. Nickel-63 would also be present in the components initially. However, it has a comparatively short radiological half-life (100 yr) relative to the predicted transport times, and it would not be found at detectable concentrations in the aquifer beneath the burial ground or at more distant downgradient locations.

The combination of the 0.1-cm/yr recharge rate and a single component source term resulted in the lowest groundwater concentrations, whereas a high recharge rate with the 120 component array produced the highest concentrations. The time required for nickel to reach the unconfined aquifer was 68,000 years or more after disposal. These times are comparable to the period over which the burial site's current geological and hydrological configuration is expected to be altered substantially by global climate changes and other geohydrological processes.

The potential radiation dose from nickel isotopes was estimated for maximally exposed individuals (MI) using well water 100 m and 5000 m down-gradient from the disposal site, and for an individual who uses river water downstream from the groundwater entry point. A potential collective dose over ten thousand years was also determined for the downstream population that might use river water for domestic purposes, recreation, and irrigation. The MI exposure was evaluated using a family farm scenario, in which the individual was assumed to use well water for domestic purposes as well as for irrigation of a 2-hectare farm. The individual was assumed to produce 100% of his vegetables and animal products on the farm, and to spend half time outside exposed to external irradiation from radionuclides deposited on the soil. A scenario was also evaluated for a person who uses river water for drinking, whose diet consists of food produced on land irrigated with river water, who eats fish caught in the river, and who uses the river for recreational purposes.

In evaluating the collective dose from river use, the future population was assumed to drink water from the river and to obtain food from local sources that use river water for irrigation. The population was assumed to

remain at an equilibrium level of 3 million people over the 10,000 years of the analysis. The analysis was conducted for an exposure beginning at the time of peak ^{59}Ni concentration in the river and assumed that the concentration remained constant throughout the 10,000-year period.

The potential annual dose to the MI using a well either 100 m or 5000 m downgradient from the burial site was less than 0.001 mrem/yr from all pathways. Consumption of drinking water accounts for no more than 3×10^{-4} mrem/yr of the total. The dose to an individual using river water was less than 2×10^{-10} mrem/yr, and the collective population dose over 10,000 years was less than 5.3×10^{-4} person-rem.

The potential annual dose to the MI on a family farm can be compared to the dose received by an individual in the local area from natural background radiation today, which results in an annual per capita effective dose equivalent of about 300 mrem/yr (NCRP 1987). Therefore, the maximum dose to a hypothetical individual from ^{59}Ni disposed at the burial ground is a small fraction of the natural background dose.

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1.0 INTRODUCTION

This study was performed at Pacific Northwest Laboratory^(a) to estimate the quantities of nickel radioisotopes that may be released to groundwater following the permanent disposal of large carbon steel components containing nickel alloys and lead at the Hanford Site 218-E-12B Burial Ground. Previous analyses for shallow waste disposal sites at Hanford (U.S. Department of Energy [DOE] 1987) indicated that use of groundwater from a downgradient well to maintain a two-hectare family farm resulted in the maximum individual exposure to radionuclides.

Estimates of nickel migration in groundwater required information concerning the geologic structure underlying the burial trench, the geochemical properties of the soil/groundwater system, and the physical-hydraulic properties of the site. This study was divided into four technical tasks as follows:

- characterization of the geologic structure and physical-hydraulic properties of soil strata underlying the burial site
- characterization of the geochemical system for release and migration of nickel from the disposal site, including potential competitive effects of lead on soil adsorption of nickel
- modeling of water flow and nickel migration in soils and groundwater to determine future concentrations in surface- and groundwater at downgradient locations
- modeling radiation dose to individuals and populations that may use contaminated surface and groundwater for domestic, agricultural, and recreational purposes in the future.

Information needed to perform the assessment was obtained from existing literature and databases where possible, and laboratory studies were conducted to provide data that were not available from other sources. Methods used to obtain necessary data and the results of the transport analysis are described in the following sections.

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2.0 THE STRATIGRAPHY, GEOHYDROLOGIC PROPERTIES, AND SUBSURFACE FLOW CONCEPTUAL MODEL OF THE GEOHYDROLOGIC SYSTEM UNDERLYING THE 218-E-12B BURIAL GROUND

This section of the report describes the geologic and hydrologic characteristics of the 218-E-12B Burial Ground, which is located in the 200-East area of the Hanford Site (Figure 2.1). In general, the geology of the area consists of sedimentary deposits of varying thickness, known as the Hanford formation, overlying a relatively impermeable basalt or clay formation. The suprabasalt geohydrology in the vicinity of the burial ground consists of unsaturated and saturated flow within the unconsolidated sediments of the Hanford formation. In general, the Hanford formation in the vicinity of the burial ground is characterized by a relatively thick (up to 100 m) unsaturated zone and thin (0 to tens of meters) saturated zone. The general stratigraphy of the area is discussed in this section, followed by a description of an existing conceptual model for the movement of water through this geohydrologic system. A more detailed description of the burial ground geology is presented in Appendix A of Rhoads et al. (1992).

2.1 STRATIGRAPHY

Three principal stratigraphic units are present beneath the 218-E-12B Burial Ground (from oldest to youngest): 1) the Miocene Columbia River basalt, interbedded with 2) sedimentary deposits of the Ellensburg Formation, and 3) the glaciofluvial Hanford formation (Last et al. 1989). The Hanford formation (informal name) makes up the vadose zone and the unconfined aquifer directly beneath the 218-E-12B Burial Ground. The Hanford formation is the principal post-basalt sedimentary unit beneath the burial ground, where it averages about 61 m in thickness (Figure 2.2). The Hanford formation was deposited during periods of cataclysmic flooding, which occurred intermittently during the last ice age (Pleistocene Epoch). The last flood event took place approximately 13,000 years ago (Mullineaux et al. 1978).

FIGURE 2.1. Location Map of the 218-E-12B Burial Ground Study Area

FIGURE 2.2. Hydrogeologic Cross Section Beneath the 218-E-12B Burial Ground

The Hanford formation is divided into three sediment types, principally on the basis of texture: 1) gravel-dominated, 2) sand-dominated, and 3) silt-dominated (Figure 2.2). Although all three sediment types are present within the 218-E-12B Burial Ground, the gravel-dominated sediments predominate. Heterogeneity among these sediment types within the Hanford formation occurs at such a scale that strata identified in one borehole often cannot be correlated to those in adjacent boreholes; this is partially a result of the complex history of flooding in the area (Last et al. 1989).

2.2 PHYSICAL PROPERTIES OF THE HANFORD FORMATION

Representative data for physical properties of the Hanford formation components are currently limited. Some limitations of physical property data include 1) a bias in favor of the finer-grained sediments (e.g., sand, silt, and clay) because representative samples of coarse-grained sediments are difficult to collect and measure and 2) physical properties that may be significantly altered during sampling (e.g., density increases and hydraulic conductivity decreases due to compaction during coring). For these reasons, *in situ*, field-tested measurements of physical properties are believed to be more representative than those obtained in the laboratory, although in practice they may be difficult to obtain. Additionally, samples collected from outcrops are more likely to reflect *in situ* conditions than are borehole samples. The geologic characteristics of the Hanford formation beneath the burial site were determined from previous studies of these soils, as well as from samples taken for this study from the faces of the burial pit and nearby boreholes. Where good field data were available in the literature, they were used for this assessment. In the absence of such information, the study employed data from laboratory analyses of collected samples. The physical and hydraulic properties of Hanford sediments are summarized in Table 2.1.

The physical properties of the Hanford formation vary depending on sediment type and depth below ground surface. Sediments at greater depth have lower hydraulic conductivities and porosities, and higher densities because of diagenetic effects such as compaction and recrystallization. Gravel-dominated sediments tend to have higher saturated hydraulic conductivities and densities

TABLE 2.1. Physical and Hydraulic Properties of Hanford Formation Sediments in the Vicinity of the 218-E-12B Burial Ground

PROPERTY	VADOSE ZONE SEDIMENTS				SATURATED ZONE SEDIMENTS
	Silt-Dominated	Sand-Dominated	Gravel-Dominated	Undifferentiated	
Saturated Hydraulic Conductivity (m/day)					
• Laboratory Tested	0.15-4.05 ^(a)	1.0-9.4 ^(a)	3-95 ^(b)	9-25 ^(b) 0.3-30 ^(a)	Not Available
• Field Tested	Not Available	Not Available	Not Available	Not Available	25 ^(c) 450-27,500 ^(d) 600-3050 ^(e)
Unsaturated Hydraulic Conductivity (m/day)	Not Available	Not Available	Not Available	Not Applicable	Not Applicable
Density (g/cm ³)	1.4-1.6 ^(a)	1.3-1.7 ^(a)	1.8-2.0 ^(a)	Not Applicable	1.8-2.0 ^(a)
Total Porosity	0.3-0.5 ^(a)	0.3-0.5 ^(a)	0.2-0.4 ^(a)	Not Applicable	0.2-0.4 ^(a)
Effective Porosity	Not Applicable	Not Applicable	Not Applicable	Not Applicable	0.1 ^(f)
Moisture (wt%)	5-30 ^(a)	1-5 ^(a)	1-5 ^(a)	Not Applicable	Not Available
Moisture (vol.%) (saturated)	Not Applicable	Not Applicable	Not Applicable	Not Applicable	20-30
Percent Gravel	0	0-60	60-100	Not Applicable	(g)
Percent Sand	0-50	50-100	0-60	Not Applicable	(g)
Percent Silt	50-100	0-50	0-20	Not Applicable	(g)
Percent Clay	0-10	0-5	0-5	Not Applicable	(g)

Sources: (a) Laboratory results obtained for this project

(b) Bjornstad (1990)

(c) Borghese et al. (1990)

(d) Last et al. (1989)

(e) Graham et al. (1981)

(f) Bierschenk (1959)

(g) Values used are assumed to be the same as those for gravel-dominated sediments in the vadose zone.

(1.8 to 2.0 g/cm³), as well as lower total porosity (0.2 to 0.4) and moisture (1 to 5 wt%) compared with fine-grained sediments. The gravel-dominated sediments are usually more poorly sorted and display characteristic large-scale foreset bedding, with dips up to 30 degrees from horizontal. Maximum clast size ranges from pebbles to boulders greater than 1 m in diameter. Gravel-dominated sediments often grade upward into sand- and silt-dominated sediments. The sand-dominated sediments are intermediate in most properties between gravel-dominated and silt-dominated sediments (Table 2.1), and are better sorted than the gravel-dominated sediments. Structurally, the sand-dominated sediments display planar, subhorizontal laminations ranging from a few millimeters (mm) to centimeters (cm) in thickness.

Silt-dominated sediments (predominantly silt- to fine sand-sized particles) have relatively low saturated hydraulic conductivities (0.15 to 4.05 m/day) and densities (1.4 to 1.6 g/cm³). Silt-dominated sediments compose a relatively small proportion of the Hanford formation within the 218-E-12B Burial Ground. However, they are significant because they tend to concentrate and control vadose-zone transport of moisture as a result of their higher moisture retention capacity (up to 30 wt%). Two thin (a few feet or less) silt-dominated beds are present in the upper 15 m (50 ft) of the exposed faces at the 218-E-12B Burial Ground (Figure 2.2). Additional silt beds may be present at depths below 15 m, but they are difficult to identify from drilling logs because of their limited thickness and higher densities that result from compaction at these depths.

2.3 HYDROLOGY

Groundwater in the vicinity of the 218-E-12B Burial Ground occurs under both unconfined and confined conditions. In this report, the term "groundwater" refers to any moisture in soil below the surface zone where evaporation and transpiration by plants can occur. The focus of this report, however, is on the vadose zone (the unsaturated sediments between the soil surface and the water table) and the unconfined aquifer (the uppermost saturated zone bounded by the vadose zone above and a relatively impermeable confining layer below, see Figure 2.2). Groundwater at the Hanford Site also

exists under confined conditions (bounded above and below by relatively impermeable rock or soil layers); however, the communication of the confined aquifers with infiltrating moisture from the soil surface is relatively limited compared with that in the unconfined aquifer.

The unconfined aquifer occurs within the Hanford formation, and is generally ≈ 3 m thick in the vicinity of the burial ground under current conditions. Columbia River basalt forms a relatively impermeable confining layer beneath the Hanford formation. Point measurements of the saturated thickness of the unconfined aquifer beneath the 218-E-12B Burial Ground range from ≈ 0.6 to 1.3 m. However, the unconfined aquifer is thicker both north and south of the burial ground. Aquifer tests within gravel-dominated sediments of the Hanford formation indicated that saturated hydraulic conductivities in these Hanford soils ranged from 25 to 27,500 m/day (Table 2.1). The highest hydraulic conductivity values measured at the Hanford Site were associated with matrix-depleted bouldery gravels, which are generally not present beneath the 218-E-12B Burial Ground. In contrast, the saturated hydraulic conductivity of basalt underlying the unconfined aquifer is less than 1 m/day in the horizontal direction and 3×10^{-5} m/day in the vertical direction (Lum et al. 1990).

To the south of the 200-West and 200-East Areas, the regional groundwater flow in the unconfined aquifer is currently from west to east. The groundwater flow direction in this area is controlled primarily by recharge from the Cold Creek and Dry Creek drainages (located to the southwest of the Site), by the artificial recharge mound associated with B Pond, and by the Columbia River. The B Pond consists of a series of unlined, interconnected waste water disposal ponds that receive effluent from the 200-East Area. Discharges to B Pond will be discontinued as activities at the Hanford Site shift to emphasize cleanup; therefore, the present conditions reflecting B Pond operation are not expected to persist. Based on the December 1987 water table shown in Figure 2.3, groundwater in the vicinity of the 218-E-12B Burial Ground is moving from B Pond toward the west, turning north through the gap between Gable Butte and Gable Mountain. Although this general flow pattern would be expected to persist over the near future, the exact level of

FIGURE 2.3. Presumed Groundwater Gradients and Flow Directions in the Unconfined Aquifer Beneath the Hanford Site 200 Areas (Data represent the aquifer status as of December 1987, including a contribution from B-Pond recharge.)

the water table is subject to short-term fluctuations caused by seasonal variations in precipitation and changes in site operations (Woodruff and Hanf 1992).

For this assessment, two cases of regional groundwater flow in the unconfined aquifer have been simulated assuming the absence of B-Pond recharge--one representing arid climate conditions (0.5 cm/yr recharge) and another illustrating potentially more humid climate conditions (5 cm/yr recharge). Water table contours for these two cases are shown in Figures 2.4 and 2.5. A conceptual model describing the unconfined aquifer beneath the 200-East Area under these conditions is discussed in a subsequent section of this document and in Jacobson and Freshley (1990).

S9209014.2

FIGURE 2.4. Water Table Contours for the Steady-State 0.5-cm/yr Recharge Case (Graph scales in feet; contour lines in feet above mean sea level.)

S9209014.3

FIGURE 2.5. Water Table Contours for the Steady-State 5.0-cm/yr Recharge Case (Graph scales in feet; contour lines in feet above mean sea level.)

3.0 GEOCHEMISTRY OF NICKEL IN SOIL/GROUNDWATER SYSTEMS AT THE HANFORD SITE

This portion of the study investigated the solubility and adsorption characteristics of nickel in Hanford sediments that exist at the burial site. Possible competitive effects of other materials in the components on soil adsorption of nickel were also investigated as part of the study. This was considered because the presence of other metals could reduce adsorption of nickel in the soil column and thereby accelerate its transport to downgradient groundwater locations. Iron and chromium (as steel alloys) and lead metal are also significant constituents in the components, and of the three only lead is sufficiently soluble in a chemical form that could compete with nickel for soil adsorption sites. Therefore, this investigation included experiments to determine whether lead might influence adsorption of nickel in Hanford sediments. Estimates of nickel release rates from the buried components, and removal by geochemical processes as it moves with soil water through the subsurface, depend upon corrosion rates for the various alloys in the components, the chemical form of released metals, their mobility, and their probable interactions with aqueous geochemistry and sediment mineralogy beneath the site.

As part of this task, a literature search and geochemical modeling using the MINTEQA2 computer code were performed to determine solubility constraints for nickel compounds in Hanford groundwater. A set of laboratory experiments was also conducted to determine the solubility of nickel compounds in Hanford groundwater and to verify the accuracy of the MINTEQA2 solubility calculations for subsurface conditions at the burial ground. Batch adsorption studies were conducted to determine the distribution coefficients (R_d) for nickel on Hanford sediments, as well as to investigate potential competitive effects of lead on nickel adsorption. Flow-through column experiments were performed to confirm the results of the batch experiments.

3.1 METHOD FOR DETERMINING NICKEL SOLUBILITY AND NICKEL ALLOY CORROSION RATES

Nickel and lead will be present in the burial ground as nickel-bearing alloys and solid lead metal, respectively. Groundwater transport predictions require knowledge of the rate at which the metals leach into solution as well as their soil adsorption properties as discussed in the following sections. For this analysis, leaching of nickel from the metal components would result from corrosion of the metal alloys and subsequent dissolution of the corrosion products in subsurface water. Because there are several nickel-bearing alloys in the components, either process (corrosion or dissolution) could control the amount of nickel released to the subsurface environment depending on the relative corrosion rates of the alloys and the solubility of the resulting corrosion products. The primary nickel-bearing alloys in the components include HY-80 steel, Type 304 stainless steel, and Inconel Alloy 600. Estimates for corrosion rates of these alloys were obtained from a literature search, the results of which are reported by Jenkins (1993).

Type 304 stainless steel and Inconel Alloy 600 would be expected to resist corrosion, because these alloys form surface films that are generally protective under conditions typical of those found in the Hanford subsurface environment. Based on the literature review, the corrosion rate for Type 304 stainless steel was estimated to be less than $0.02 \text{ mg/dm}^2\text{-yr}$. This upper-end corrosion rate was used to bound nickel releases in this analysis. The corrosion rate of Inconel 600 was assumed to be 70% of that for 304 stainless steel ($0.01 \text{ mg/dm}^2\text{-yr}$ when rounded to 1 significant figure for modeling). For these alloys, the corrosion rate was assumed to control nickel release, and dissolution of corrosion products was assumed to occur immediately after their formation.

In the case of HY-80, which forms a relatively nonprotective corrosion layer, the corrosion rate in the burial ground environment was estimated to be less than $70 \text{ mg/dm}^2\text{-yr}$. The upper-end corrosion rate for HY-80 was substantially higher than for the other alloys. Because of its higher corrosion rate in soil, release of nickel from the HY-80 corrosion products was assumed to be controlled by a thermodynamic solubility process. That is, it was assumed that recharge water percolating through the burial ground would contact the

buried components and immediately dissolve enough corrosion product to achieve equilibrium with the nickel compounds that would be thermodynamically stable under ambient conditions. This approach ignored any kinetic and mass transfer hindrances to dissolution that are likely to lower the concentration of nickel in leachate from the buried components and, therefore, provides a theoretical upper bound for the flux of nickel into groundwater.

The solubility of nickel in Hanford groundwater was estimated in laboratory experiments where nickel compounds were equilibrated in a simulated Hanford groundwater solution. The results of the laboratory analyses were confirmed with MINTEQA2 version 3.0 (Allison, Brown, and Novo-Gradac 1991), an updated version of the MINTEQ geochemical equilibrium code (Felmy et al. 1984). The composition of groundwater used in the MINTEQA2 calculations and the laboratory experiments was based on analysis of water taken from well 6-S3-25, the results of which are listed in Table 3.1. This well is located on the Hanford Site, approximately 17 km from the 200 East Area, where the unconfined aquifer is relatively uncontaminated by liquid discharges from operating facilities. Based on the well water analysis shown in Table 3.1 and assuming equilibrium with the solid phases of plausible nickel compounds, the MINTEQA2 code was used to calculate equilibrium solution concentrations for these compounds.

The MINTEQA2 modeling and the results of earlier laboratory experiments (Mattigod, Rai, and Felmy, in press)^(a) indicated that the thermodynamically stable nickel compounds in Hanford groundwater could be either hydroxides or carbonates. Based on this information, additional laboratory studies were conducted to determine the solubility of nickel in Hanford groundwater for both of these compounds. The solubility of nickel in Hanford groundwater was evaluated by measuring the concentrations of nickel in equilibrium with nickel hydroxide $[\text{Ni}(\text{OH})_3]$ and nickel carbonate $[\text{Ni}_3\text{CO}_3(\text{OH})_3]$. Three samples each of nickel hydroxide and nickel carbonate were equilibrated with Hanford groundwater. About 300 mg of solids were equilibrated with 30 ml of groundwater in

(a) Mattigod, S. V., D. Rai, and A. R. Felmy. In press. "Hydrolysis Constant for Ni(2) and the Solubility Product of Crystalline $\text{Ni}(\text{OH})_2$." J. Soln. Chem.

TABLE 3.1. Chemical Composition of Hanford Groundwater^(a) (Well 6-S3-25) used for Input to the MINTEQA2 Computer Code for Determination of Solubility for Nickel Compounds

Analyses	Sample Used (4/90)	Range in Composition (1985-1990)
Bulk Chemistry:		
pH	8.14	7.82 - 8.47
Eh(mv)	309	283 - 385
Cations (meq/L) ^(b)	5.29	4.9 - 5.4
Anions (meq/L)	4.60	4.6 - 6.1
Individual Constituents: Concentration, mg/L		
Al	<0.03	<0.03 - 0.064
B	0.1	0.02 - 0.10
Ba	0.08	0.04 - 0.08
Ca	48.8	48.8 - 58.8
Cd	<0.004	<0.004
Cr	<0.020	<0.02 - 0.034
Cu	<0.004	<0.004
Fe	<0.005	<0.005 - 0.008
K	9.9	4.9 - 9.9
Li	<0.004	<0.004
Mg	14.6	13.2 - 14.6
Mn	<0.002	<0.002 - 0.13
Na	32.1	23.8 - 32.1
P	<0.1	<0.1
Pb	<0.06	<0.06
Si	16.4	14.6 - 16.4
Sr	0.25	0.23 - 0.25
Zn	<0.02	<0.02 - 0.08
Fluoride	0.5	<0.5 - 0.7
Chloride	27	21. - 27.
Nitrite	<0.3	<0.3
Nitrate	<0.5	<0.5
Phosphate	<0.4	<3.
Sulfate	75	63. - 92.
Total Alkalinity (as CO ₃ ²⁻)	67.5	67.5 - 92.4
Total Organic Carbon	1	0.3 - 1.7

- (a) From: Serne, R. J., V. L. LeGore, C. W. Lindenmeier, J. A. Cambell, and J. L. Conca. 1991. Progress on Task D Performance Assessment Support in the Solid Waste Technology Support Project. Pacific Northwest Laboratory, Richland, Washington.
- (b) One equivalent (eq) equals the gram-atomic mass of a given cation divided by its ionic charge.

a 50 ml centrifuge tube by shaking at room temperature. Aqueous samples were withdrawn at 14, 28, 50, and 123 days and filtered with 0.0018- μ m membrane cones. The equilibrium concentrations of nickel in samples from the experiments were determined using inductively coupled plasma-mass spectrometry (ICP-MS).

3.2 DESCRIPTION OF ADSORPTION EXPERIMENTS

The following sections describe procedures used to prepare soil samples for laboratory analyses and to conduct adsorption studies. Batch tests were performed to provide data for transport modeling, and a flow-through column test was conducted to confirm the results of batch tests.

3.2.1 Preparation and Analysis of Hanford Sediment Samples

Approximately 45 kg of sediment representative of the most common material in the pit wall strata were collected near the base of the burial trench. This material weighed 43 kg after air drying, and 29 kg of finer soil remained after screening out the coarser (>2 mm) material. The 14 kg of coarse gravel was discarded and the "fine" material was used in all further characterization and laboratory studies to determine the geochemical fate of nickel. The "fines" included sand, silt, and clay, which were thoroughly mixed prior to characterization and laboratory studies. The silt and clay size fractions of the "fines" were characterized using X-ray diffraction (XRD) to determine their mineralogy. Analyses for calcium carbonate content, total cation exchange capacity, organic carbon, and particle size were also performed.

3.2.2 Batch Adsorption Tests

The primary metals of interest in this study were nickel (including the radioisotopes ^{59}Ni and ^{63}Ni) and lead. The distribution coefficients (R_d) for nickel were measured both in the absence and presence of lead using batch adsorption tests, and the batch R_d values were corroborated using flow-through column tests. The experimental details for performing both batch and column adsorption tests are described in Relyea et al. (1980), Relyea (1982), and Serne and Relyea (1983). The specific experiments performed for this study included batch adsorption tests using fines from samples collected at the

burial trench, and a representative Hanford groundwater (from well 6-S3-25) spiked with nickel at initial concentrations ranging from 4.0×10^{-7} M ($24 \mu\text{g/L}$) to 5.0×10^{-5} M ($2900 \mu\text{g/L}$). The potential competitive effects of lead on adsorption of nickel were studied by varying the nickel concentrations in the presence of an initial lead concentration of 1.0×10^{-6} M (0.21 mg/L). The lead concentration was chosen to approximate the solubility limit for lead in Hanford groundwater, based on the results of a previous study (Rhoads et al. 1992), allowing a margin of safety to prevent possible precipitation of lead compounds in the test system. The solution-to-solid ratio used in the batch tests was approximately 30 mL:1 g, with contact times of 7, 10, or 26 days. Solution was separated from soil by centrifugation followed by filtration through a $0.22\text{-}\mu\text{m}$ membrane.

The nickel concentrations in solution were determined after separation of the soil using two different analytical techniques. Some batch tests contained ^{63}Ni tracer that was measured by liquid scintillation counting, whereas other tests contained only stable nickel, which was measured in solution by ICP-MS. In both types of experiments, the quantity of nickel adsorbed onto the sediment was assumed to equal the difference in mass or activity between the influent and effluent solutions. The value of R_d for nickel in these batch adsorption experiments was then estimated using Equation (3.1).

$$F_d = \frac{C_{\text{inf}} - C_{\text{eff}}}{C_{\text{eff}}} \frac{V}{W} = \frac{C_{\text{soil}}}{C_{\text{solution}}} \frac{V}{W} \quad (3.1)$$

C_{inf} = nickel concentration or tracer activity in influent solution (g/mL or counts/mL)

C_{eff} = nickel concentration or tracer activity in effluent solution (g/mL or counts/mL)

C_{soil} = nickel concentration or tracer activity in the soil (g solute/g soil or counts tracer/g soil)

C_{solution} = nickel concentration or tracer activity in solution (g/mL or counts/mL)

V = volume of solution used (mL)

W = weight of soil used (g).

3.2.3 Column Adsorption Tests

A flow-through column experiment was performed to corroborate the R_d values calculated for nickel using the batch adsorption experiments. In order to obtain data in a timely manner, the flow-through column test was conducted at a relatively rapid flow rate (approximately 10 pore volumes/day). Therefore, the residence time for groundwater in the column test was only 2.4 hours, which is extremely short compared with the predicted residence times of recharge water in the vadose zone sediments at Hanford (50 to 2150 yr, see Section 4.2). Conditions in the flow-through column test thus represent extremes that are likely to over-predict nickel mobility based on adsorption kinetics and colloidal transport potential. The extremely short residence time in the laboratory column test could potentially prevent some soil adsorption that would normally occur at lower flow rates. The kinetic energy imparted to the system as a result of the higher flow rate used in the column tests might also promote colloidal transport to a greater extent than would be expected under field conditions. Therefore, the column test should represent worst-case conditions for nickel adsorption, in addition to providing a check on whether other processes not adequately covered by batch adsorption tests are important in this geochemical system.

The column test permits direct determination of the retardation factor (R_f) from the number of pore volumes of effluent required to reach a breakthrough ratio, C_{eff}/C_{inf} , of 50%. A 24-mL sediment column was contacted with Hanford groundwater that had been spiked with stable nickel at a concentration of approximately 0.59 mg/L in addition to ^{63}Ni tracer. Effluent solutions were analyzed using the same methods used in the batch experiments. From the observed breakthrough curve and retardation factor, the R_d values can be calculated using Equation (3.2)

$$R_f = 1 + \rho_s [1 - \eta]/\eta R_d \quad (3.2)$$

where R_f = the retardation factor v_w / v_n (velocity of water + velocity of solute)

ρ_s = particle density of the soil material (mass/unit volume)

η = porosity (dimensionless)

and R_d = distribution coefficient (mL/g)

because R_f is measured and ρ_s and η are obtained during the test set up.

3.2.4 Adsorption Isotherm Models

The results of a suite of experiments evaluating the effect of solute concentration on adsorption while other parameters are held constant are called an "adsorption isotherm." Two adsorption isotherm models used frequently are the Langmuir and Freundlich models. In this analysis, an isotherm model was used to represent nickel adsorption in the subsurface environment because it more realistically describes adsorption as a function of solute concentration than does a linear model that uses a single value for the distribution coefficient at all concentrations.

The Langmuir model has been used to describe adsorption of gas molecules onto homogeneous solid surfaces (crystalline materials) that exhibit one type of adsorption site (Langmuir 1918). Many investigators have extended the Langmuir adsorption model to describe adsorption from solution onto solid adsorbates, including heterogeneous solids. The Langmuir model for adsorption is

$$X = \frac{bX_m C}{1 + bC} \quad (3.3)$$

where X = amount of solute adsorbed per unit weight of solid

b = a constant related to the energy of adsorption

X_m = maximum adsorption concentration of the adsorbate

C = equilibrium solution concentration of the adsorbate.

Substituting $1/B$ for b , we obtain

$$X = \frac{X_m C}{B + C} \quad (3.4)$$

A plot of values of X (y-axis) versus values of C (x-axis) passes through the origin and is nearly linear at low values of C . As C increases, X should approach X_m . One can rearrange Equation (3.4) by taking its reciprocal and multiplying both sides by $X \cdot X_m$, to yield $X = -B(X/C) + X_m$. Then by plotting X on the y-axis and (X/C) on the x-axis, it is possible to determine the value for $-B$ from the slope of the best-fit line and the value of X_m from the intercept.

For radionuclide adsorption onto heterogeneous soils and sediments, the Langmuir model is typically a poor predictor of actual adsorption events, although Salter, Ames, and McGarrah (1981a) cite several instances where the Langmuir isotherm has successfully fit adsorption of trace solutes on natural substrates. Further, Salter, Ames, and McGarrah (1981b) discuss recent modifications of the Langmuir model to accommodate two distinct sites and competition of two adsorbates (the nuclide and the ion it replaces on the adsorbent), which should further extend this conceptual model's usefulness on natural substrates. However, for more complex natural systems, the Langmuir model is of limited utility.

Because of the complexity of the subsurface environment at Hanford, the Freundlich isotherm model was considered to be more appropriate than the Langmuir for incorporation into the transport analysis. The Freundlich isotherm model (Freundlich 1926) is defined as

$$X = KC^N \quad (3.5)$$

where X = amount of solute adsorbed per unit weight of solid

C = equilibrium solute solution concentration

K, N = constants.

The Freundlich model does not account for finite adsorption capacity at high solute concentrations; however, the current experiments were conducted

under conditions that were not expected to exceed the adsorption capacity of the sediment component. The Freundlich isotherm can be transformed to a linear equation by taking the logarithms of both sides of Equation (3.5):

$$\log X = \log K + N \log C \quad (3.6)$$

When $\log X$ is plotted on the y-axis and $\log C$ on the x-axis, the best-fit straight line has a slope of N , and $\log K$ is its intercept. When $N = 1$, the Freundlich isotherm represented by Equation (3.5) reduces to a linear relationship, and because X/C is the ratio of the quantity of solute adsorbed to the equilibrium solution concentration (the definition of R_d), the Freundlich K is equivalent to the value of R_d .

Because adsorption isotherms at very low solute concentrations are often linear, either the Freundlich isotherm or the Langmuir isotherm (when the product bC is much smaller than 1) can be used to fit data at low solute concentrations. The value of N for the adsorption of many radionuclides is often significantly different from 1. In this case, the value of R_d is not constant, but is a function of the solute concentration.

Both isotherm models can be compared to data from experiments that systematically vary the mass of a trace constituent while holding all other parameters as constant as possible. It is important to consider the total mass of the element present, including all stable and radioactive isotopes, when evaluating isotherms. It is incorrect to calculate isotherms based on a single radioactive tracer if the system includes several isotopes (both stable and radioactive) for a particular element. For convenience, isotherm experiments generally consider only the total concentration or radioactivity for a given constituent, combining all isotopes present in the system.

The isotherm concept is a step up in sophistication over the constant distribution coefficient (R_d) model. It must be stressed that isotherm models as expressed by Equations (3.4), (3.5), and (3.6) explicitly consider dependency of the distribution coefficient only on the concentration of the contaminant of interest in a solution. Isotherm models do not explicitly consider other solid and solution parameters that can influence adsorption.

In this study, the potential for lead to influence nickel adsorption was assessed in batch adsorption experiments, the results of which were incorporated into the isotherm model used to evaluate groundwater transport.

The incorporation of isotherm adsorption models into transport codes is relatively easy. Each of the aforementioned isotherm equations can be rearranged to calculate a distribution coefficient, R_d , that is a function of C , the solution concentration of the contaminant, and one or two constants. As the transport model solves for C , substitution of an equation that depends only upon C (and derivable constants) for the R_d in the retardation factor should be straightforward. For simple cases, analytical closed-form solutions are possible, or numerical approximation methods can be used. Thus, with little additional work or computer storage requirements, most transport codes can be formulated to predict radionuclide migration with an isotherm adsorption model. In this analysis, the Freundlich isotherm was chosen for incorporation to the subsurface transport model because it is most appropriate for use under field conditions where more than one type of adsorption site is expected to exist for the solute of interest.

3.3 RESULTS OF GEOCHEMISTRY STUDIES

The following sections present results of the characterization of Hanford sediments in the vicinity of the 218-E-12B Burial Ground, the laboratory experiments to determine the solubility of nickel in Hanford groundwater, estimates of nickel corrosion rates, and the extent of adsorption of nickel on these sediments in the presence and absence of lead.

3.3.1 Sediment Characterization

The sediment characterization data are shown in Tables 3.2 through 3.6. Table 3.2 shows the particle size distribution of the "fines" used in all lab tests and the *in situ* sample taken from the field. The particle size distribution was determined by combining dry sieving and a modified pipet method (using a centrifuge) to separate silt from clay. The dry sieving showed that most of the material was between 1 and 2 mm in diameter. Texturally the sediment would be considered a very coarse sand. The particle density of the fines used in laboratory testing was 2.84 g/cm³, similar to most Hanford

TABLE 3.2. Particle Size Distribution of Hanford Formation Sediments in the Vicinity of the 218-E-12B Burial Ground

<u>Sediment Type/Size</u>	<u>Percent by Weight</u>	
	<u>Fines Used in Adsorption Studies</u>	<u>In Situ Composition of Vadose Zone Sediments</u>
Gravel (>2mm)	0.00	32.7
Sand (<2mm to >63 μ m)	99.55	67.0
Silt (<63 μ m to >2 μ m)	0.39	0.26
Clay (<2 μ m)	0.06	0.04

TABLE 3.3. Saturation Paste pH and Composition of Soil from the 218-E-12B Burial Ground^(a)

<u>Composition</u>	
<u>Constituent</u>	<u>Concentration (μg/g dry soil)</u>
PO ₄ ³⁻	< 0.07
Cl ⁻	6.3
NO ₃ ⁻	0.9
SO ₄ ²⁻	16.4
Total Alkalinity (as CO ₃ ²⁻)	14.52
Na ⁺	5.26
Mg ²⁺	1.97
Ca ²⁺	8.10
Sr ²⁺	0.04
Si	3.98
Dissolved Organic Carbon	1.66

(a) From: Serne, R. J., V. L. LeGore, C. W. Lindenmeier, J. A. Cambell, and J. L. Conca. 1991. Progress on Task D Performance Assessment Support in the Solid Waste Technology Support Project. Pacific Northwest Laboratory, Richland, Washington.

TABLE 3.4. Amorphous Oxide, Calcium Carbonate, and Organic Carbon Content of the "Fines" in Soil from the 218-E-12B Burial Ground

<u>Oxide</u>	<u>Percent by Weight</u>
SiO ₂	0.41
Al ₂ O ₃	0.26
Fe ₂ O ₃	0.41
MnO ₂	0.02
CaCO ₃	0.82
Total Organic Carbon	≤ 0.01

TABLE 3.5. Exchangeable Cations and Total Cation Exchange Capacity for Soil from the 218-E-12B Burial Ground

<u>Cation</u>	<u>Concentration (meq/100 g)</u>
Na ⁺	0.12 ± 0.02
Ca ²⁺	4.15 ± 0.63
Mg ²⁺	0.87 ± 0.03
Total Exchangeable Cations	5.14 ± 0.6
Total Cation Exchange Capacity	5.27 ± 1.2

TABLE 3.6. Mineral Content of the Clay Size Fraction of the "Fines" Used in Adsorption Experiments

<u>Mineral</u>	<u>Percent by Weight</u>
Illite	13
Talc	Not Detected
Horneblende	1
Kaolinite	3
Chlorite	2
Vermiculite	8
Smectite	38
Quartz	9
Plagioclase	26

sediments. The saturated paste pH and saturation extract composition are given in Table 3.3. Table 3.4 shows the percent of amorphous hydrous oxide, calcium carbonate, and total organic carbon in the fines. Exchangeable cation results are given in Table 3.5, and the mineralogy of the clay fraction of the sediment is shown in Table 3.6.

The exchangeable cation and total cation exchange capacity analyses suggest that coarse sand has an exchange capacity of about 5 meq/100 g, a value typical of sandy Hanford sediments. In fact, given the rather coarse texture of the sediments, values as low as 3 meq/100 g might be expected. The clay size fraction contains crystallites of plagioclase feldspar, quartz, and the clays smectite, illite, vermiculite, and kaolinite. Smectite predominates in the layer silicates, as is generally found for Hanford sediments.

3.3.2 Solubility of Nickel Compounds in Hanford Groundwater

Estimates of nickel solubility in Hanford groundwater were obtained both from a geochemical computer model and from laboratory experiments that were intended to simulate the subsurface environment at the burial ground. Calculations using MINTEQA2 indicated that the solubility of nickel in Hanford groundwater should be controlled by $\text{Ni}(\text{OH})_2$, and the solubility of Ni^{2+} was predicted to be between 10 and 20 mg/L. However, results of the laboratory experiments indicated that the actual solubility of nickel hydroxide in Hanford groundwater was lower than the MINTEQA2 predictions.

Results of the laboratory analyses for nickel solubility are presented in Figure 3.1, which indicates that nickel hydroxide is about an order of magnitude less soluble than nickel carbonate. Based on these results, nickel hydroxide was determined to control the solubility of nickel in Hanford groundwater. Nickel hydroxide appears to have reached equilibrium within approximately 50 days at a concentration of 1.9 ± 0.3 mg/L. The solubility of nickel carbonate peaks at a concentration of 30 mg/L approximately 50 days into the experiment and then declines. X-ray diffraction analysis of the solids suggests that this behavior may result from slow conversion of nickel carbonate to a more stable crystalline phase.

FIGURE 3.1. Solubility of Nickel Hydroxide and Nickel Carbonate in Hanford Groundwater

Geochemical modeling was conducted using the aqueous geochemical equilibrium code MINTQA2 to determine if available thermodynamic data would confirm the empirical solubility results for nickel hydroxide in Hanford groundwater. Major ion concentrations used in the input file were determined from chemical analysis of well water at the Hanford Site (see Table 3.1), with carbonate concentrations adjusted to reflect equilibrium concentrations in the laboratory experiments. Initial results using default thermodynamic data from the software database estimated the solubility of nickel hydroxide at 17 mg/L (Rhoads et al. 1992).

Subsequently, the solubility constant for nickel hydroxide was modified from that in the original MINTQA2 database to reflect recent results obtained by Mattigod, Rai, and Felmy (in press).^(a) In addition, it appears that the MINTQA2 database default stability constant for the formation of the dissolved complex species NiCO_3^0 may be too large. The log K for the formation

(a) Mattigod, S. V., D. Rai, and A. R. Felmy. In press. "Hydrolysis Constant for Ni(2) and the Solubility Product of Crystalline $\text{Ni}(\text{OH})_2$." J. Soln. Chem.

of NiCO_3^0 is 6.87 in the MINTEQA2 database, whereas results from a carbonate-oxalate stability constant correlation indicate that the stability constant is more likely to be 5.37 (Byrne, Kump, and Cantrell 1988). MINTEQA2 calculations using the revised solubility constant for nickel hydroxide and the lower stability constant for NiCO_3^0 predicted the solubility of nickel in Hanford groundwater to be 13 mg/L. Both of the MINTEQA2 solubility estimates for nickel in Hanford groundwater were substantially higher than the results of the laboratory studies. However, because of uncertainty about the applicability of data in the MINTEQA2 database to the Hanford subsurface environment, the empirical determination of 1.9 mg/L from the laboratory studies is judged to be more reliable.

3.3.3 Batch Adsorption Studies

The first suite of batch adsorption experiments was conducted for a 7- to 10-day equilibration period, the results of which are presented in Figures 3.2 and 3.3. Figure 3.2 illustrates the 7- to 10-day R_d values for nickel determined in both the radioactive tracer and stable nickel experiments as a function of the final equilibrium nickel concentration in solution. These experiments were conducted both in the presence and absence of an initial lead concentration of 1×10^{-6} M (0.21 mg/L). The values for R_d in both sets of experiments ranged from 48 to 340 mL/g, which resulted in an estimate of 45 mL/g when extrapolated to the solubility limit for nickel. In these experiments, R_d decreased as the concentration of nickel in the equilibrated solution increased, which was consistent with expectations because the relative adsorption of nickel should decrease as more adsorption sites on the soil are occupied.

The effect is also illustrated in Figure 3.3, where nickel adsorption data are plotted as an adsorption isotherm. When the final concentrations of nickel in solution are plotted against those in soil, the concentration of nickel in the soil increases in a linear fashion with the concentration in solution (i.e., total adsorption increases). As can be seen from both Figures 3.2 and 3.3, the effect of lead on nickel adsorption was negligible compared with experimental variation in the data.

FIGURE 3.2. Distribution Coefficients (R_d) for Nickel from Radiotracer and Stable Nickel Experiments, 7- to 10-Day Equilibration Period

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FIGURE 3.3. Nickel Adsorption Isotherm for 7- to 10-Day Equilibration Period

Data from the 7- to 10-day radiotracer batch tests, with and without added lead, were fitted to a Freundlich isotherm for use in the groundwater transport modeling. Because the results of these tests (Figure 3.3) indicated that soil adsorption was linear with respect to the logarithm of the solution nickel concentration, the nickel concentrations used in the tests were low enough that the adsorption capacity of the soils was not exceeded. Therefore, the conditions for modeling the Freundlich isotherm as described in Section 3.2.4 were met. The form of the isotherm is described in Equation (3.5), with constants determined by the results of the batch tests. For this set of experimental data, the Freundlich isotherm model results in the following equation:

$$X = 332 * C^{0.775} \quad (3.7)$$

where X = amount of solute adsorbed per unit weight of solid

C = equilibrium solute concentration

and 332, 0.775 = constants obtained from results of batch tests.

The 26-day batch test results are shown in Figure 3.4 and are compared with 7- to 10-day results for the radiotracer studies. Overall, the R_d estimated in 26-day experiments appears to be more variable, but the values are generally comparable to those obtained in the 7- to 10-day studies. Because infiltration rates in Hanford sediments are extremely low, the 7- to 10-day R_d values should be conservative estimates of nickel adsorption under field conditions.

3.3.4 Column Adsorption Study

A column adsorption experiment was conducted to confirm the validity of the batch adsorption results in a flow-through system, the results of which are shown in Figure 3.5. The influent concentration of Ni^{2+} was 0.59 mg/L, and the experiment continued for 90 days, at which time approximately 1000 pore volumes of solution had passed through the column. In contrast to the behavior that was expected based on the batch adsorption studies (shown by the solid curve in Figure 3.5), the effluent concentration peaked at about 60% of

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FIGURE 3.4. Distribution Coefficients (R_d) for Nickel from Radiotracer Experiments, Comparison of Values for 7- to 26-day Tests

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FIGURE 3.5. Results of Nickel Column Adsorption Experiments

the influent concentration after 500 pore volumes and stabilized at a slightly lower concentration thereafter. Attempts to explain the behavior of this system using a kinetic model (the expected results of which are also shown in Figure 3.5) did not reproduce the observed breakthrough curve.

The likely explanation for the behavior of nickel in this study is that its solubility was actually lower than predicted by the MINTEQA2 model, resulting in precipitation of nickel compounds from solution as it flowed through the column. Results of experiments to determine the solubility of nickel compounds in Hanford groundwater supported this hypothesis. Despite the anomalous behavior of nickel in this test system, the results indicated that no unforeseen mechanisms that could significantly accelerate nickel transport were operative during the study period. The transport of nickel through the column was actually lower than predicted based on the distribution coefficients observed in the batch experiments.

3.4 DISCUSSION AND RECOMMENDATIONS FOR TRANSPORT MODELING

Geochemical parameters, as determined in these laboratory studies, were recommended as input to the nickel transport model. The solubility limit for nickel in Hanford groundwater was determined to be in the range of 10-20 mg/L based on the results of the MINTEQA2 geochemical model. Laboratory experiments to determine the solubility of potential nickel corrosion products in Hanford groundwater indicated a lower value of about 2 mg/L for nickel hydroxide, which is expected to be the stable corrosion product in this subsurface environment. Because of uncertainty concerning the applicability of thermodynamic data in the MINTEQA2 database to the Hanford environment, the 2 mg/L solubility determined in experiments under site-specific conditions was recommended for use in the nickel transport modeling.

The results of batch adsorption studies indicated that R_d values for nickel ranged from 50 mL/g near the solubility limit of nickel in Hanford groundwater to 340 mL/g for equilibrium solution concentrations of nickel. The data set obtained from 7- to 10-day radiotracer batch adsorption studies was chosen for modeling the Freundlich isotherm because it is the most conservative of the experimental results and is subject to the least variability.

The resulting isotherm model described by Equation 3.7 was recommended to model soil adsorption in the groundwater transport analysis.

4.0 WATER FLOW AND TRANSPORT OF NICKEL IN HANFORD SOILS AND GROUNDWATER

In order to estimate rates of groundwater flow and nickel transport at the 218-E-12B Burial Ground, a conceptual model was developed for the unconfined aquifer underlying the Hanford Site using available data on geohydrologic properties and geochemistry (as described in previous sections of this report). This section describes a simulation technique used to analyze the migration of nickel from components disposed at the 218-E-12B Burial Ground and presents the results of the transport analysis.

Metal components in shallow land burials will be subjected to degradation by the natural environment, primarily through chemical degradation and dissolution by infiltrating water. The resulting leachate will move downward until it enters the unconfined aquifer, where it will disperse in the groundwater and be transported to the Columbia River. The consequences of nickel migration will ultimately depend on the amount of nickel in the disposal area, the rate at which it is leached from the buried components, and interactions of nickel in solution with the soil and water chemistry.

The conceptual model for the burial site was the basis for estimated rates of groundwater movement through the vadose zone and the unconfined aquifer, and for predicted rates of nickel migration from the burial ground to downgradient locations and to the Columbia River. The Coupled Fluid, Energy, and Solute Transport (CFEST) code (Gupta et al. 1982; 1987) was used to produce a two-dimensional model of the regional aquifer in order to obtain parameters necessary for the nickel transport analysis. A modified version of the TRANSS code (Simmons et al. 1986) was then employed to simulate mass flow and transport through the vadose zone and the unconfined aquifer using a one-dimensional streamtube approach. This approach is similar to that used in previously published documents for the Hanford Site (DOE 1987; 1989), except that the modified TRANSS code utilizes a variable adsorption isotherm that depends on groundwater concentration rather than a constant value for the distribution coefficient. Simulations were performed for a single component, and for a 4 x 30 array of 120 components.

Parameters of interest in this investigation, which were incorporated into the simulations, included the following:

- consequences of burial for either a single component, or for 120 components in a 4 x 30 array
- three recharge rates (representing recharge through an engineered barrier, or through an unprotected burial site under either arid or humid climate conditions)
- release of nickel to the subsurface environment – based on high-end estimates for corrosion rates of Type 304 stainless steel and Inconel 600, or a best estimate solubility for the nickel corrosion product in the case of the less corrosion-resistant material (HY-80)
- a variable nickel distribution coefficient based on the Freundlich adsorption isotherm
- three receptor locations (wells entering the unconfined aquifer at locations 100 m and 5000 m from the burial site, and the Columbia River).

Various combinations of these parameters were considered in the analysis to provide relatively conservative estimates for nickel release and transport.

For disposal of a maximum of 120 components, the quantities of stable and radioactive nickel available for release and transport were based on the maximum total quantity of these materials expected to be present at the site. In the analysis for a single component, the estimated quantities for the array were assumed to be distributed equally among the 120 components potentially stored there. An average component was assumed to contain 121,000 kg HY-80 steel (3.5% Ni by weight), 41,000 kg Type 304 stainless steel (12% Ni by weight), and 67,000 kg Inconel 600 (80% Ni by weight). The surface areas associated with these alloys in a single component were 101,000 dm², 33,000 dm², and 52,000 dm², respectively. The radioactive nickel content associated with each alloy in a single component was estimated to be 6.6 Ci ⁵⁹Ni for Type 304 stainless steel and 209 Ci ⁵⁹Ni for Inconel 600. The HY-80 is not radioactive.

Nickel concentrations in the unconfined aquifer were estimated for receptors using water from a downgradient well located 100 m or 5000 m from the burial site, and the total annual flux of nickel to the Columbia River was

also predicted. A groundwater well located 100 m downgradient of the burial ground represents the presumed boundary of the disposal site when the government no longer maintains institutional control (DOE 1990). The groundwater well located 5000 m downgradient represents the location of the family farm scenario that has been evaluated in previous environmental impact statements (EISs) (DOE 1987; 1989).

Recharge rates of 0.1, 0.5, and 6.0 cm/yr through the burial ground were modeled for this analysis. The 0.1-cm/yr recharge rate represents an estimate of the maximum infiltration that would be expected through a multilayered soil/rock barrier designed to control infiltration over the waste site, as described in a recent EIS for the Hanford Site (DOE 1987). The 0.5-cm/yr recharge rate, applied to both the burial ground and the Hanford Site regional aquifer model, was chosen to represent the present relatively arid climate. The potential effect of a more humid climate on nickel transport in the future was modeled using a 6-cm/yr recharge rate for the burial ground and a 5-cm/yr recharge rate for the Hanford regional aquifer model.

4.1 ONE-DIMENSIONAL STREAMTUBE APPROACH

A one-dimensional streamtube approach as implemented in the TRANSS code was described by Simmons et al. (1986). A detailed discussion of the streamtube approach and its application was presented in the EIS for disposal of Hanford defense wastes (DOE 1987); a similar approach was used for this study. This approach assumes transport along a series of streamlines that form a streamtube. Advection was assumed to be the dominant process for nickel movement, and transverse dispersion was assumed to be negligible. Advection is defined as contaminant transport in the downgradient direction at the average groundwater velocity, whereas transverse dispersion is transport perpendicular to the direction of the average velocity caused by variations in velocity about the average. Neglecting transverse dispersion in the streamtube resulted in conservation of mass along its entire length from the burial ground to the river.

The streamtube approach of Simmons et al. (1986) is capable of simulating streamtubes that begin in the vadose zone and end at the river, as is

expected to be the case for nickel migration from the 218-E-12B Burial Ground. Water was assumed to infiltrate vertically through the burial site and to dissolve nickel in the components en route to the water table. During transport within the unconfined aquifer, leachate entering the water table from the disposal site was assumed to mix in the upper portion of the aquifer. A mixing depth of 2.5 m was assumed for the 0.5 cm/yr recharge regional model in this analysis, which was a departure from the 5-m mixing depth used in previous analyses (DOE 1987). It was employed because of the shallow depth of the unconfined aquifer in the vicinity of the 218-E-12B Burial Ground. A 5-m mixing depth was assumed for the high recharge (5 cm/yr) regional aquifer model.

Nickel transport was modeled within the streamtube in the unconfined aquifer to downgradient locations and ultimately to the regional aquifer discharge area at the river. Some portion of the dissolved nickel may chemically adsorb to the sediments that constitute the geologic units through which the streamtube passes, and a Freundlich adsorption isotherm was incorporated into the model to simulate this process, as described in Section 3.0. The following sections describe the conceptualization of the vadose zone and the unconfined aquifer underlying the 218-E-12B Burial Ground, and the rates of water movement through each. The transport of nickel to downgradient locations was then estimated for several combinations of key input parameters to the TRANSS computer code.

The TRANSS code applied for this study is a less sophisticated transport model than multi-dimensional transport codes such as VAM3D and PORFLO-3 because it is one-dimensional and omits transverse dispersion. The results from this less sophisticated code have not been directly compared with those of multi-dimensional models.

4.2 VADOSE ZONE WATER FLOW

Sedimentary deposits situated between the ground surface and the water table constitute the vadose zone (Figure 2.2). Within these deposits, water only partially fills the available pore space between grains of silt, sand, and gravel, and is generally assumed to drain vertically downward until it

enters the unconfined aquifer or until it contacts a low permeability layer of clay or rock. Significant amounts of water that infiltrate the surface soils may be held at the surface and evaporate if the soil cover is sufficiently fine-grained. Infiltrating water may also be transpired if the surface is covered with vegetation. Any remaining water is assumed to constitute recharge to the underlying aquifer. Fine-grained sediments at or near the soil surface tend to hold water at the location where evapotranspiration is greatest; coarse-grained sediments permit the water to drain to a depth where these forces are less effective or nonexistent.

The long-term average precipitation rate at Hanford is 17 cm/yr. However, the recharge rate to the unconfined aquifer at any specific location may vary widely depending on the nature of the soil surface and the presence or absence of vegetation. Experimental results indicate that the infiltration rate may range from less than 1% of annual precipitation beneath a vegetated surface of silt loam soil to approximately 80% of the annual precipitation rate beneath a surface of bare gravel without vegetation (Smoot et al. 1989). This corresponds to a range of infiltration rates from ≤ 0.2 to 13 cm/yr for the current Hanford climate. A long-term change in the precipitation rate would likely produce a corresponding change in the recharge rate through all types of soil surfaces.

For the purposes of this analysis, a recharge rate of 0.5 cm/yr was used to model infiltration through the undisturbed vadose zone based on the current relatively arid climate. The 0.1 cm/yr and 6 cm/yr recharge rates were used to model infiltration through the burial ground under conditions that would either delay or accelerate nickel transport through the vadose zone, respectively, compared with the current undisturbed situation. The 0.1 cm/yr recharge rate corresponds to the estimated maximum infiltration through a protective, multilayered infiltration and intrusion barrier that is being considered for several waste disposal sites at Hanford. The 6 cm/yr recharge rate represents a situation where infiltration is increased either by a long-term change to a wetter climate, or where alteration of the soil surface permits greater retention of incident precipitation.

In modeling the regional unconfined aquifer, an average site-wide recharge rate of 0.5 cm/yr was used to represent its condition under the current climate, and a recharge rate of 5 cm/yr was used to represent the aquifer in a wetter climate scenario (Evans et al. 1988; Jacobson and Freshley 1990). When modeling transport from the burial ground to the receptor locations, the 0.5 cm/yr regional aquifer model was coupled with either the 0.1 cm/yr or the 0.5 cm/yr infiltration rate through the vadose zone. The 6 cm/yr infiltration rate was coupled with the wet climate regional aquifer model. The assumptions employed in this analysis are consistent with those used for previous Hanford Site studies (DOE 1987; DOE 1989; Golder Associates, Inc. 1991), and are based on the range of results observed in field studies on the Site (Gee et al. 1992; Graham et al. 1981; Isaacson and Brown 1978; Kirkham and Gee 1984; Last, Easley, and Brown 1976).

Stratigraphic control for the geohydrology in the vicinity of the burial site was obtained from two boreholes (299-E34-7 and 299-E35-1) located immediately adjacent to the burial ground excavation (Figure 2.2). Sediments below the 218-E-12B Burial Ground are composed primarily of gravel-dominated sediments mixed with variable amounts of sand and silt. Gravels are predominantly in the pebble-to-cobble size range, but boulders are occasionally present. The vadose zone is approximately 60 m (200 ft) thick in the vicinity of the 218-E-12B Burial Ground, of which approximately 15 m (50 ft) has been excavated for disposal of the metal components. The thickness of the vadose zone is similar for all simulations because the unconfined aquifer beneath the burial ground is less than 3 m thick in the vicinity of the burial ground, even in the presence of artificial recharge from current site operations (see Section 2.3). The resulting distance from the base of the disposal area to the bottom of the vadose zone is approximately 45 m (150 ft). This analysis did not account for the presence of backfill around the buried components; all water and leachate flow through the 45-m thickness of vadose zone was assumed to begin at the bottom level of the excavation. Therefore, neither the distribution of water through the upper 15 m of the soil profile nor the time required for it to flow through that area have been evaluated as part of this analysis.

The physical properties of sediment samples collected from the faces of the 218-E-12B Burial Ground pit were analyzed in the laboratory; the results are summarized in Table 2.1. Samples could only be analyzed from those portions of the geologic profile without large cobbles or boulders; therefore, analytical data are biased in favor of the finer-grained sediments. Measurements of saturated hydraulic conductivity for these samples (i.e., sand- and silt-dominated) ranged from approximately 0.3 to 30 m/day (1 to 100 ft/day). The soil sample descriptions and saturated hydraulic conductivity data were used to match the significant geologic units in the vadose zone to moisture-retention data measured previously for samples from the Hanford formation in the vicinity of the burial ground (Bjornstad 1990). Because of the larger pebbles, cobbles, and boulders in many horizons, the measured samples were assumed to be representative of the lowest conductivity samples in the geologic section.

Moisture-retention and saturated hydraulic-conductivity data from well 299-E33-30 at the 42.4- to 42.7-m (139- to 140-ft) depth interval were assumed to represent the Hanford formation in the area (Bjornstad 1990). This well is located in the 200-E Area, approximately 2 km west of the 218-E-12B Burial Ground. Samples from this location had the highest measured saturated hydraulic conductivity of any of the samples – approximately 95 m/day. However, it is expected that there are strata within the vadose zone with conductivities above and below this value. The value of 95 m/day is approximately 4 times the saturated hydraulic conductivity of 25 m/day obtained from an aquifer test within the saturated zone at well 299-E34-7 (Borghese et al. 1990), immediately adjacent to the burial ground (see Figure 2.2). It is important to note that the vadose zone consists of a sequence of units that contain a range of hydraulic properties. The 299-E33-30 sample was assumed to be representative of the overall profile. The validity of this interpretation is limited by the technical difficulty of measuring the physical and hydraulic properties of the coarser-grained units, which would be expected to have the highest values. The highest hydraulic conductivity values measured at the Hanford Site, hundreds to thousands of meters per day (Last et al. 1989), are probably associated with

matrix-depleted bouldery gravels, which are generally not present beneath the 218-E-12B Burial Ground.

Water travel times through the vadose zone were calculated using characteristic curves developed for Hanford sediments, assuming that unit gradient conditions prevail beneath the burial ground. Soil moisture characteristic curves describe the relationships between water content, water potential, and unsaturated hydraulic conductivity. The unsaturated hydraulic conductivity values were calculated using the method of van Genuchten (1978; 1985). These values were based on the measured saturated hydraulic conductivity (95 m/day) and the water content versus water potential data for well 299-E33-30. The resulting characteristic curves, plotted in Figure 4.1, indicate that increasing water content results in increasing unsaturated hydraulic conductivity values up to the limiting value of saturated hydraulic conductivity. For unit gradient conditions in the unsaturated zone, this recharge flux was assumed to equal the unsaturated hydraulic conductivity. The curves were used to determine the water content that results from the unsaturated hydraulic conductivity corresponding to the assumed recharge rates of 0.1, 0.5, and 6.0 cm/yr. The water content was then used to calculate the pore-water velocity through the 45-m-thick vadose zone beneath the 218-E-12B Burial Ground. Table 4.1 shows the water content, pore-water velocity, and vadose zone travel time for selected fluxes through the 46-m-thick vadose zone based on 299-E33-30 well data (Bjornstad 1990). The estimated groundwater travel times are 50 yr for the 6-cm/yr recharge case, 480 yr for the 0.5-cm/yr recharge case, and 2,150 yr for the 0.1-cm/yr case.

4.3 GROUNDWATER FLOW MODELING

The CFEST code (Gupta et al. 1982; 1987) was applied to model groundwater flow in the unconfined aquifer and to generate streamlines and travel times used in the transport simulations. The CFEST code is being considered for acceptance as a Hanford Site standard for constructing models of the unconfined aquifer. Evans et al. (1988) describe selection of the CFEST code for application to the unconfined aquifer at the Hanford Site. Development

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FIGURE 4.1. Moisture Retention Curves for Soil Samples from Well 299-E33-30
(Volumetric water content = water volume ÷ total volume;
water potential = hydraulic head in cm H₂O)

TABLE 4.1. Volumetric Water Content, Pore-Water Velocity, and Water Travel Time for Three Recharge Rates Through the Burial Ground

	Recharge Rate (cm/yr)		
	0.1	0.5	6.0
Volumetric Water Content	0.047	0.052	0.066
Pore-Water Velocity (cm/yr)	2.1	9.6	91.
Water Travel Time (yr)	2150	480	50

and calibration of the CFEST model is described by Evans et al. (1988) and Jacobson and Freshley (1990).

The conceptual model of the aquifer on which the CFEST model relies is based on information available in the early to mid-1980s. At that time, the top of the basalt formation in the vicinity of the 218-E-12B Burial Ground was believed to lie above the water table. Consequently, it was assumed that the Hanford formation was not saturated beneath the burial ground. As a result of this and other recent studies, the basalt is known to exhibit a saddle-shaped topography, with a trough running northwest to southeast underlying the 218-E-12B Burial Ground (see Figure 2.3). A similar topography would be expected to exist under any condition of relatively high local recharge, whether from long-term climate change or site operations.

The water level in this part of the unconfined aquifer has risen during the past 10 years because of increased liquid discharges to B Pond, which is located southeast of the burial ground. When the discharges to B Pond are discontinued after decommissioning of facilities in the 200-East Area, the water table in this region will drop and the basalt layer that underlies the Hanford formation may extend above the saturated zone (Figure 2.4). In the conceptual model of groundwater flow beneath the burial ground it was assumed that infiltrating water and leachate passing through the vadose zone would contact the basalt that extends above the water table, flow laterally across the basalt surface, and then enter the unconfined aquifer at the nearest location to the south of the discharge point.

4.3.1 Groundwater Travel Times

This section summarizes the groundwater travel times for the unconfined aquifer that were calculated using the CFEST model. Groundwater flow in the unconfined aquifer was modeled to determine the average groundwater velocity at each recharge rate and the corresponding travel times in the aquifer to wells 100 m and 5000 m from the burial ground, and to the Columbia River. Steady-state flow fields were simulated for site-wide recharge rates of 0.5 and 5.0 cm/yr (see Figures 4.2 and 4.3), which were intended to represent the range of long-term climatic conditions expected at the Hanford Site (Jacobson and Freshley 1990).

FIGURE 4.2. Finite-Element Grid for the CFEST Model of the Unconfined Aquifer at a 0.5-cm/yr Recharge Rate

FIGURE 4.3. Finite-Element Grid for the CFEST Model of the Unconfined Aquifer at a 5.0-cm/yr Recharge Rate

Recharge conditions at specific locations on the Hanford Site may vary significantly from these site-wide averages depending on land use and vegetation cover. Installations with engineered infiltration barriers would be expected to exhibit the lowest long-term recharge rates. A recharge rate of 6 cm/yr was used to represent infiltration through the 218-E-12B burial ground under maximum recharge conditions in the absence of a protective barrier. As noted previously, this is not substantially different from the site-wide recharge rate of 5 cm/yr, and provides consistency with studies already completed (Golder Associates, Inc. 1991). Travel times to downgradient locations were determined using an effective porosity of 0.1 with the modeled pore-water velocity in the unconfined aquifer. The effective porosity value of 0.1 (10%) was determined in a previous study of aquifer characteristics (Bierschenk 1959) and has been used for most subsequent model applications involving the unconfined aquifer at the Hanford Site (e.g., DOE 1987).

Streamlines from the burial ground to the river were generated using the CFEST aquifer model for each regional recharge rate considered in this analysis (Figures 4.4 and 4.5). Because the 1980s conceptual model of the aquifer predicted that Hanford formation sediments beneath the burial ground would be unsaturated, streamlines in the transport model could not begin directly below the burial ground. As a result, it was assumed that the difference in hydraulic conductivity between the sedimentary formation and the underlying basalt would cause water infiltrating downward through the vadose zone to flow across the top of the basalt until it reaches the unconfined aquifer. The basalt formation beneath the burial ground slopes toward the south. Therefore, several locations to the south of the burial ground were selected for the transport model as probable points where infiltrating water could enter the unconfined aquifer.

Groundwater flow in the unconfined aquifer for the site-wide 0.5-cm/yr case would be primarily from west to east across the entire Hanford Site, with streamlines intersecting the river at points along the site's eastern boundary (Figure 4.4). The lower water table (Figure 2.4) would also cause more of the

FIGURE 4.4. Streamlines from the 218-E-12B Burial Ground to the Columbia River for the 0.5-cm/yr Regional Recharge Case

FIGURE 4.5. Streamlines from the 218-E-12B Burial Ground to the Columbia River for the 5.0-cm/yr Regional Recharge Case

basalt to be unsaturated. The permeability of the basalt is several orders of magnitude lower than that of the overlying Hanford formation sediments. In basalt to be unsaturated. The permeability of the basalt is several orders of magnitude lower than that of the overlying Hanford formation sediments. In the 0.5-cm/yr site-wide recharge case, most of the basalt beneath the 200-East Area would lie above the water table. Under these conditions, water infiltrating through the burial ground was assumed to flow toward the south, along the interface between the Hanford formation sediments and the low permeability basalt formation, until it reaches the aquifer and enters a streamtube flowing toward the east or southeast (Figure 4.4). The streamlines (and travel times) are modeled beginning at points near the southern boundary of the 200-East Area. Table 4.2 shows estimated travel times for three locations along each streamline based on the CFEST aquifer model for a site-wide recharge rate of 0.5 cm/yr. Average groundwater travel times in the unconfined aquifer were 0.57 yr to the 100-m well, 59 yr to the 5000-m well, and 190 yr to the river.

Water table contours for the site-wide 5.0-cm/yr recharge case (Figure 2.5) indicate that the water table would be higher than for present-day conditions. This would cause a groundwater divide to develop in the low-gradient area south of Gable Mountain. The streamlines originating in the 200-East Area show that the groundwater flow direction in the unconfined aquifer would be generally northward toward the river through the gap between Gable Butte and Gable Mountain (Figure 4.5). Travel-time values for three locations along each streamline for a 5.0-cm/yr recharge rate are provided in Table 4.3. Average groundwater travel times in the unconfined aquifer were 0.51 yr to the 100-m well, 2.4 yr to the 5000-m well, and 29 yr to the river.

Aquifer simulations for the two regional recharge rates predicted markedly different groundwater flow patterns in the unconfined aquifer. This occurs because the regional recharge rate determines both the hydraulic gradients within the unconfined aquifer and the level of the water table with respect to the basalt that underlies the aquifer. This especially affects the estimated travel times to the 5000-m well and to the Columbia River. At the 0.5-cm/yr recharge rate, bedrock beneath the burial ground is exposed and nearby groundwater is predicted to flow toward the east or southeast.

TABLE 4.2. Travel Times Along Individual Streamlines in the Unconfined Aquifer for the 0.5-cm/yr Site-Wide Recharge Case

<u>Groundwater Travel Time to Downgradient Location (Yr)</u>			
<u>Streamline No.</u>	<u>100-m Well</u>	<u>5000-m Well</u>	<u>Columbia River</u>
1	0.57	67	180
2	0.66	81	180
3	0.86	110	200
4	0.52	56	180
5	0.52	47	190
6	0.42	33	230
7	0.52	37	230
8	0.48	44	160
Average:	0.57	59	190

TABLE 4.3. Travel Times Along Individual Streamlines in the Unconfined Aquifer for the 5.0-cm/yr Site-Wide Recharge Case

<u>Groundwater Travel Time to Downgradient Location (Yr)</u>			
<u>Streamline No.</u>	<u>100-m Well</u>	<u>5000-m Well</u>	<u>Columbia River</u>
1	0.63	1.8	34
2	0.95	2.0	25
3	0.12	2.4	35
4	0.25	3.3	32
5	0.14	2.7	26
6	0.96	2.1	24
Average:	0.51	2.4	29

Groundwater travel in the unconfined aquifer to the 100-m well takes between 0.42 and 0.86 yr; however, the flow rate at downgradient locations becomes much slower as the hydraulic gradient decreases. At the 5.0-cm/yr recharge rate, bedrock beneath the burial site is saturated, and groundwater is ultimately predicted to flow northward through the gap between Gable Butte and Gable Mountain. The travel time in the unconfined aquifer to a 100-m well in this case is between 0.12 and 0.96 yr. However, the groundwater flow rate increases at distant points because it is subject to higher hydraulic gradients and hydraulic conductivities as it reaches the 5000-m well and discharges to the Columbia River.

In general, groundwater travel times through the unconfined aquifer are less affected by recharge rate than are the rates of water infiltration through the vadose zone. Travel times in the unconfined aquifer to the 100-m well for individual streamtubes reflect the local hydraulic gradients and the conductivity of the sediments through which the streamtube passes. The longer travel times for some of the streamtubes in the 5.0-cm/yr recharge case probably reflect the relatively high hydraulic heads to the north of the burial ground that would tend to resist groundwater flow in that direction. The hydraulic gradients to the south and east are lower and provide less resistance to flow at the 0.5-cm/yr recharge rate. Average travel times to the 100-m well predicted by the unconfined aquifer model for both recharge rates were virtually identical – 0.51 yr and 0.57 yr – for the high- and low-regional recharge cases, respectively.

4.3.2 Groundwater Velocity

The results of the two-dimensional flow model for the unconfined aquifer provide critical input to the one-dimensional TRANSS transport model calculations, because the estimated groundwater travel times are used to predict the groundwater (i.e., pore-water) velocity at the 100-m well, the 5000-m well, and the river. The groundwater velocity in the unconfined aquifer is critical to the transport calculation because it defines the volume of water available for dilution of nickel leaching from the burial ground. Therefore, the resulting nickel concentration in the unconfined aquifer will be inversely proportional to the calculated groundwater velocity; that is, a higher groundwater velocity will result in lower nickel concentrations in the unconfined aquifer, and a lower groundwater velocity will result in higher nickel concentrations in the unconfined aquifer.

Because the one-dimensional TRANSS code could not accommodate the variable groundwater velocities predicted by the CFEST model at different distances from the burial ground, it was necessary to choose a single value to represent the groundwater velocity in the unconfined aquifer at all locations in order to achieve a fully consistent model. Average velocities for the 0.5- and 5.0-cm/yr recharge rates at the 100-m well were similar, at 175 and 197 m/yr, respectively. These values were also within the range predicted for

the 5000-m well, which was 115 m/yr for a recharge rate of 0.5 cm/yr and 2073 m/yr for the 5.0-cm/yr case. Thus, the average velocity at the 100-m well, or 186 m/yr, was used as input for the TRANSS code in all cases. This value was consistent with the existing CFEST model of the Hanford Site, and it should produce a reasonably conservative estimate of nickel concentration in the aquifer for the high-recharge case.

4.3.3 Limitations of the CFEST Groundwater Flow Model

The regional Hanford Site CFEST model generally reproduces the behavior of the unconfined aquifer at Hanford under current conditions, particularly for areas where the saturated zone is thicker. However, the CFEST model does not have sufficient resolution to accurately predict local-scale behavior of the aquifer immediately beneath the 218-E-12B Burial Ground. Three conditions of the unconfined aquifer are described in this document: the current state of the aquifer (including recharge from site operations), its state under an enduring low site-wide recharge rate (0.5 cm/yr) in the absence of artificial recharge, and its state under an enduring high site-wide recharge rate (5 cm/yr). In the vicinity of the burial ground, groundwater in the unconfined aquifer currently flows west from B Pond, turning north through the gap between Gable Butte and Gable Mountain (see Figure 2.3). The simulated 5.0-cm/yr recharge rate model also predicts a similar flow pattern. An evaluation of the regional geology for this study indicated that relatively high hydraulic heads were measured north of the burial ground. Based on those hydraulic head measurements, flow in the area north of the burial ground appears to be toward the southeast, into the burial ground, rather than toward the northwest (Graham et al. 1984; Rhoads et al. 1992, Appendix A).

The low-recharge simulation predicts desaturation of basalt beneath the burial ground, in which case infiltrating water would contact the basalt layer above the water table and be diverted toward the south. Newcombe et al. (1972) documented the unconfined aquifer prior to the advent of significant artificial recharge resulting from Hanford operations, indicating that the pre-Hanford groundwater flow in the vicinity of the 200 Areas was generally from west to east, from the Yakima Ridge-Umtanum Ridge highland toward the Columbia River. Streamlines drawn in the vicinity of the 200 Areas form a

streamtube with this general characteristic (Bjornstad 1990). A similar flow field might be expected to return with the decommissioning of artificial recharge sources at Hanford, such as the B Pond and U Pond. A more complete description of the Hanford Site hydrologic model can be found in Rhoads et al. (1992) and Jacobson and Freshley (1990).

Using the present version of the CFEST model, it is difficult to predict the exact direction of groundwater flow in the unconfined aquifer beneath the burial ground for the hypothetical conditions evaluated in this study. Even if the CFEST model were modified to increase its resolution in the region of the burial ground, uncertainty concerning future recharge rates limits the accuracy of predicted groundwater flow patterns at distant future times. The west-to-east flow direction is considered most probable for the 0.5-cm/yr recharge case, whereas the south-to-north direction is more likely for the 5.0-cm/yr recharge rate (see Figures 4.4 and 4.5). Nevertheless, these regional models suffice to provide estimates of nickel transport for the conditions assumed in this analysis.

4.4 NICKEL TRANSPORT

Nickel transport was estimated with the TRANSS code according to the general methodology described in Section 4.1. For this analysis, the code was modified to incorporate a function for the distribution coefficient that varies with groundwater concentration. The Freundlich adsorption isotherm described in Section 3.3.3 was chosen as the most appropriate model for the subsurface conditions at the burial ground, and this model was added to the code as an option for calculating the distribution coefficient. Recharge rates of 0.1 cm/yr, 0.5 cm/yr, and 6 cm/yr were used for the analysis. These recharge rates were intended solely to represent flow through soil beneath the burial ground. The regional recharge rates of 5 cm/yr and 0.5 cm/yr used for the groundwater flow modeling described in Section 4.3 represent a range of values for the Hanford Site as a whole based on a likely range for future climatic cycles, native vegetation, and human impact on soil surfaces across the Hanford Site.

Based on a 1650 ft² (150 m²) storage area for each buried component, the 0.1-cm/yr recharge rate results in 150 L/yr of water that is assumed to pass through the buried components and dissolve nickel corrosion products up to the solubility limit. The 0.5-cm/yr recharge rate results in 770 L/yr of water, and the 6.0-cm/yr recharge rate results in 9200 L/yr of water contacting each component; it was assumed that this water is available to dissolve nickel compounds up to the solubility limit. Consequently, at the higher recharge rate there is a greater volume of water available to dissolve nickel from the components, and a greater quantity of nickel is released to the subsurface soil and water.

Transport simulations in the unconfined aquifer require information on the aquifer dimensions in the area of interest. Previous analyses of shallow-land disposal using the methods employed in this study (DOE 1987; 1989) have evaluated locations on the Hanford Site that were overlying the unconfined aquifer, rather than an area where the basalt formation could extend above the water table as may be the case for the 218-E-12B Burial Ground (Figure 2.3). Those studies assumed a 5-m thickness for the contaminated aquifer, which corresponded to the use of a 5-m screened interval in a downgradient well. A review of available field data (Section 2.3) estimated the thickness of the saturated zone near the burial ground at between 0.6 and 1.3 m. Therefore, it may be unreasonable to assume 5 m of saturated aquifer under low recharge conditions (i.e., the 0.5-cm/yr regional recharge rate). Because of the shallow saturated thickness of the aquifer in the vicinity of the burial ground, a smaller screened interval and an aquifer thickness of 2.5 m were more appropriate for this analysis at local recharge rates of 0.1 and 0.5 cm/yr. The 5.0-cm/yr regional aquifer model indicated that the aquifer depth would be sufficient to support a 5-m screened well, and the 5-m depth was assumed for the high local recharge (6 cm/yr) case.

Release rates of nickel to the subsurface environment were estimated in order to provide source terms for the nickel transport modeling. For HY-80 in the components, the alloy corrodes at a rate that is higher than the rate at which the corrosion products can be dissolved in groundwater; therefore, the corrosion product solubility and the assumed recharge rates determine the

maximum release rate for nickel. A solubility of 2 mg/L nickel for the most probable corrosion product (nickel hydroxide) in Hanford groundwater was recommended as described in Section 3.4 of this report. Multiplying the solubility by the annual water flux past each component yields the annual release of nickel into the soil column for each component (Table 4.4). The maximum total annual quantity of nickel released from the burial ground was estimated by multiplying the annual release per component by the number of components in the disposal array. Under equilibrium conditions, the quantity of elemental nickel entering the unconfined aquifer equals that released from the components.

Total release times for nickel from HY-80 in the buried components represent the times required to completely dissolve the entire inventory of nickel corrosion products, which were 1.4×10^7 yr, 2.8×10^6 yr, and 2.3×10^5 yr for the 0.1-, 0.5-, and 6.0-cm/yr recharge rates, respectively. For the corrosion-resistant alloys, nickel release rates are assumed to be controlled by the corrosion rate, which is independent of the groundwater recharge rate for the purposes of this analysis. In this case, the nickel release rate depends on the corrosion rate of each alloy and the surface area exposed to the environment, resulting in total release times of 6.2×10^7 yr for Type 304 stainless steel and 1.3×10^8 yr for Inconel 600. Consequently, extremely long times are predicted for complete release of elemental nickel from components in the burial ground.

TABLE 4.4. Annual Flux of Nickel into the Soil Column and the Unconfined Aquifer Beneath the 218-E-12B Burial Ground

Recharge Rate (cm/yr)	Maximum Release Rate of Elemental Nickel to the Soil Column or to the Unconfined Aquifer (grams per year)	
	Single Component	Multiple Components (4 x 30 Array)
0.1	0.31	37.
0.5	1.5	180.
6.0	18.	2200.

Release of ^{59}Ni to the subsurface environment is controlled by the corrosion rates of Type 304 stainless steel and Inconel 600. The HY-80 is not radioactive, and therefore does not contribute to the subsurface radionuclide inventory. The ^{59}Ni is assumed to be evenly distributed throughout the stainless steel and Inconel in the components; therefore, the total release time for these alloys corresponds to that of elemental nickel. However, the ^{59}Ni inventory undergoes radioactive decay and is reduced to negligible quantities over the times required for complete corrosion of these more resistant alloys in the components. Therefore, the release rate of ^{59}Ni decreases with time after burial from an initial rate of 2×10^{-6} Ci/yr.

After the metal is in solution, TRANSS models mass transport in the streamtube as described in Section 4.1 of this report. The streamtube width was based on the diagonal length through the waste disposal area (22 m for a single component and 460 m for the array of 120 components), and the streamtube height in the aquifer was based on a 2.5-m or 5.0-m mixing depth for the low and high regional recharge cases, respectively. During transport, a portion of the dissolved nickel was assumed to adsorb to soil in the geologic units underlying the burial site according to the Freundlich adsorption isotherm described in Section 3.3.3 of this report. This process would retard or delay the rate at which nickel would otherwise be transported through the vadose zone and the unconfined aquifer in comparison with the groundwater travel times. Groundwater concentrations in the unconfined aquifer were calculated at wells located 100 m and 5000 m from the burial site, and the total annual flux was calculated for discharges to the Columbia River.

4.5 NICKEL CONCENTRATIONS

The results of the transport modeling for nickel migration appear in Table 4.5, which presents the concentrations of ^{59}Ni (in pCi/L) and total elemental nickel (in mg/L) at the downgradient well locations and nickel discharges to the river (in pCi/yr or g/yr) for each source term at three different recharge rates. Although the components initially contain ^{63}Ni in addition to ^{59}Ni , ^{63}Ni would not reach the unconfined aquifer or the river in

TABLE 4.5. Peak Concentration of Elemental Nickel or ^{59}Ni in the Unconfined Aquifer or Flux to the Columbia River Following Release and Migration from the 218-E-12B Burial Ground

Receptor Location	Recharge Rate	Peak Downgradient Groundwater Concentration			
		Single Component		Multiple Components (4 X 30 Array)	
		Elemental Ni (mg/L)	^{59}Ni (pCi/L)	Elemental Ni (mg/L)	^{59}Ni (pCi/L)
100-m Well	0.1 cm/yr	3.1E-04	1.0E-15	1.7E-03	5.8E-15
	0.5 cm/yr	1.5E-03	1.2E-03	8.6E-03	6.9E-03
	6.0 cm/yr	9.2E-03	3.6E-01	5.1E-02	2.0E+00
5000-m Well	0.1 cm/yr	3.1E-04	1.9E-16	1.7E-03	1.8E-15
	0.5 cm/yr	1.5E-03	3.6E-04	8.6E-03	2.9E-03
	6.0 cm/yr	9.2E-03	1.5E-01	5.1E-02	1.1E+00
Peak River Flux					
River		Elemental Ni (g/yr)	^{59}Ni (pCi/yr)	Elemental Ni (g/yr)	^{59}Ni (pCi/yr)
River	0.1 cm/yr	3.1E-01	2.7E-12	3.7E+01	2.1E-09
	0.5 cm/yr	1.5E+00	1.9E+01	1.8E+02	8.5E+03
	6.0 cm/yr	1.8E+01	4.3E+04	2.2E+03	1.3E+07

measurable quantities because of its relatively short radiological half-life (100 yr) relative to the travel times predicted by the transport model.

For a single component source, ^{59}Ni concentrations at the 100-m and 5000-m wells ranged from 1.9×10^{-16} to 3.6×10^{-1} pCi/L, representing 3.1×10^{-4} to 9.2×10^{-3} mg/L of total elemental nickel. Discharge to the Columbia River ranged from 2.7×10^{-12} to 4.3×10^4 pCi/yr (3.1×10^{-1} to 1.8×10^1 g/yr total nickel per component). Under these conditions the maximum concentration in river water would be 4.3×10^{-10} pCi/L, (1.8×10^{-10} mg/L total nickel), based on an annual average flow rate of 1×10^{14} L/yr. The peak flux of elemental nickel is equivalent to the annual amount of nickel assumed to leach from each component (see Table 4.4), indicating that mass balance was maintained in the simulations after equilibrium conditions were established.

The maximum concentrations and discharge rates of ^{59}Ni peaked at times on the order of tens to hundreds of thousands of years (Table 4.6). Because the travel times associated with the lower recharge rates are long relative to the radiological half-life of ^{59}Ni (76,000 years), radioactive nickel concentrations decrease substantially over time compared with those of total elemental nickel. The values reported in Table 4.6 represent the time required for ^{59}Ni to travel through the vadose zone to the unconfined aquifer plus travel time in the aquifer to the indicated downgradient location.

For the component arrays, concentrations in the unconfined aquifer and releases to the river were calculated for 120 components (Table 4.5). Lower nickel concentrations at downgradient wells were predicted for the lower recharge rates because a smaller volume of water would contact the components annually. Concentrations in the unconfined aquifer ranged from 1.8×10^{-15} to 2.0 pCi/L ^{59}Ni (1.7×10^{-3} to 5.1×10^{-2} mg/L total nickel). Discharges to the Columbia River ranged from 2.1×10^{-9} to 1.3×10^7 pCi/yr ^{59}Ni (3.7×10^1 to 2.2×10^3 g/yr total nickel), resulting in a maximum surface water concentration of 1.3×10^{-7} pCi/L ^{59}Ni (2.2×10^{-8} mg/L total nickel). Plots of concentration versus time at each receptor location or release to the Columbia River for the 120 component arrays are shown in Figures 4.6 to 4.8.

TABLE 4.6. Time of Peak ^{59}Ni Concentration in the Unconfined Aquifer or Maximum Flux to the Columbia River Following Release and Migration from the 218-E-12B Burial Ground

Receptor Location	Recharge Rate	Time to Peak Downgradient Concentration (years)			
		Single Component		Multiple Components (4 X 30 Array)	
		Elemental Ni	^{59}Ni	Elemental Ni	^{59}Ni
100-m Well	0.1 cm/yr	4.0E+06	4.0E+06	4.0E+06	4.0E+06
	0.5 cm/yr	8.0E+05	8.0E+05	8.0E+05	8.0E+05
	6.0 cm/yr	7.1E+04	6.9E+04	6.9E+04	6.8E+04
5000-m Well	0.1 cm/yr	4.3E+06	4.2E+06	4.2E+06	4.1E+06
	0.5 cm/yr	9.9E+05	9.3E+05	9.4E+05	8.9E+05
	6.0 cm/yr	2.0E+05	1.6E+05	1.6E+05	1.3E+05
River	0.1 cm/yr	4.7E+06	4.7E+06	4.5E+06	4.4E+06
	0.5 cm/yr	1.3E+06	1.3E+06	1.2E+06	1.1E+06
	6.0 cm/yr	4.2E+05	3.9E+05	3.2E+05	2.9E+05

S9404021.5

FIGURE 4.6. Concentration at the 100-m Well of ^{59}Ni and Total Elemental Nickel for a 120-Component Array

FIGURE 4.7. Concentration at the 5000-m Well of ^{59}Ni and Total Elemental Nickel for a 120-Component Array

S9404021.7

FIGURE 4.8. Discharge to the Columbia River of ^{59}Ni and Total Elemental Nickel for a 120-Component Array

4.6 DISCUSSION OF GROUNDWATER MODELING

The results of this analysis are based on a relatively simplified theoretical model, and on available information about the physical and chemical environment surrounding the burial ground. There are areas of uncertainty, both in the model itself and in the parameters used by the model to produce quantitative estimates of nickel concentrations in surface- and groundwater downgradient from the burial site. An attempt has been made as part of this study to perform a credible but conservative evaluation and to determine accurate values for parameters that are critical to the analysis (nickel solubility and adsorption coefficients, for example). However, there are elements less amenable to laboratory or field analysis that influence the results obtained from the model, as discussed in this section.

One element of uncertainty in the transport calculations is the nature of the future hydrological environment, particularly the direction in which groundwater in the unconfined aquifer will flow in the absence of artificial recharge from the 200 Areas. This uncertainty is reflected in the transport model by the cross-sectional area used to represent the streamtube. The streamtube approach is a simplified representation of the water column that would transport dissolved nickel from the burial ground to the river. The model used for this analysis did not consider lateral dispersion of nickel within the streamtube; therefore, its width and depth were assumed to be constant along the entire length. The streamtube dimensions assumed for the transport model directly affected the estimated nickel concentrations in groundwater because, at a given recharge rate, they determined the volume of water into which the dissolved nickel was ultimately diluted. In this model, the depth of the streamtube was set at a constant 2.5 m or 5.0 m as appropriate for each recharge case, and the width was taken to be the longest dimension of the burial ground perpendicular to the predicted direction of groundwater flow in the unconfined aquifer. Potential widths of the streamtube for the 120-component array ranged from 61 m along the narrow axis of the burial ground to 460 m along the diagonal.

For all simulations, the diagonal dimension of the 4-by-30 array (460 m) was used for the streamtube width, which is consistent with the results of the

CFEST model (discussed in Section 4.3) and the current regional hydrology (discussed in Section 2.3), which predicts groundwater flow in the unconfined aquifer to be generally north or south and roughly perpendicular to the diagonal of the burial ground (Figure 2.3). However, the limitations of the CFEST model, described in Section 4.3.3, imply that groundwater could flow in alternate directions under some conditions. Although it is less probable over the short term than the north or south direction predicted by the current aquifer model, flow in an alternative direction would result in a higher estimated groundwater concentration for nickel because the corresponding width of the streamtube in the transport model would be reduced. A narrower streamtube would decrease the quantity of water in which leachate from the burial ground would be diluted; therefore, nickel concentrations in the unconfined aquifer would increase in proportion to the reduction in streamtube width.

The values in Table 4.5 can be used to determine the predicted nickel concentration for any groundwater flow direction by multiplying them by the ratio of the burial ground diagonal length (460 m) to the dimension perpendicular to the assumed flow direction. A flow direction from west to east along the length of the burial ground yields the maximum estimate using this procedure. In that case, the values in Table 4.5 would be multiplied by the ratio of the burial ground diagonal length (460 m) to its width (61 m), or a factor of 7.6. The ratio corresponding to any other assumed groundwater flow direction would be smaller. The estimated travel times to downgradient locations, the quantity of nickel leached from the burial ground, and the estimated maximum mass of nickel entering the Columbia River annually would not be affected because they are independent of the streamtube configuration.

Another uncertainty in this analysis is the possibility that geologic and hydrologic conditions may be transformed by natural phenomena over the very long time span required for transport of nickel to downgradient locations. DOE (1987) reported on studies by Craig^(a) and Craig and Hanson (1985) describing the potential for ice-age flooding that could affect the

(a) Craig, R. G. 1983. "Analysis of Ice-Age Flooding from Lake Missoula." Unpublished report, Kent State University, Kent, Ohio.

Hanford Site as a result of climate change over approximately the next 50,000 yr. Those studies documented evidence concerning the effects of flooding due to the catastrophic release of ice-impounded water from Lake Missoula. During the floods, it was estimated that as much as 2000 km³ of water flowed through the Pasco Basin over a period of a few weeks. A relatively weak ice age that is predicted in about 15,000 yr would not likely generate flooding of that volume; therefore, DOE (1987) did not consider glacial flooding as a potential release mechanism for hazardous wastes buried at the Hanford Site during the next 10,000 yr. However, those studies predicted a recurrence interval of 40,000 to 50,000 yr for major flooding events of this magnitude, which is comparable to the minimum times required for nickel to reach the water table in this analysis.

The possible fate of buried wastes was also examined by those studies. The first wave of a major glacial flood could scour the existing sedimentary deposits to a considerable depth; however, the sediments would likely be redeposited in the Pasco Basin. The flood waters could potentially scour out the buried components, the soil column, and portions of the aquifer as well. Any nickel in solution would also be diluted by a significant volume of flood water. With regard to disposal of radioactive wastes at Hanford, DOE (1987) concluded that "the radiological consequences of a glacial flood would not appear important in contrast to the effects of the flood itself." In this case, a similar conclusion would seem reasonable for nickel in the event of such a flood. The residual impact of nickel in the components would be expected to be insignificant compared with the effects of the flood itself.

4.7 SOURCES OF CONSERVATISM IN THE GROUNDWATER AND TRANSPORT MODELS

This evaluation was conducted using a relatively simple screening model to obtain order-of-magnitude estimates for nickel concentrations in the unconfined aquifer. As a result, the analyses of groundwater flow and nickel transport were necessarily simplified in several respects to make the simulation possible. Although the simulated groundwater flow directions, as described in sections 4.3.3 and 4.6, did not produce the most conservative results, the overall evaluation was conservative because of the nature of the

model and the assumptions made regarding other parameters. The major sources of conservatism in the analysis are described in the following sections.

4.7.1 Nickel Release Based on Solubility Limit of Nickel and Corrosion Rates

The release rates for elemental nickel in this analysis assume that all water infiltrating the soil profile associated with each buried component would come into contact with metal, and that the groundwater would reach equilibrium with the nickel corrosion products from HY-80 in the components. This results in solution entering the vadose zone at the solubility limit of nickel. In reality, groundwater contacting the components would not all be exposed to HY-80 corrosion products, nor would it necessarily reach the solubility limit for nickel in solution. Although these assumptions simplified the simulation, they resulted in higher predicted release rates and downgradient water concentrations than would actually be expected.

The corrosion rates assumed for Type 304 stainless steel and Inconel 600 also provided conservative estimates of the release rates for ^{59}Ni . The corrosion resistance of these alloys is due to formation of a protective passive layer at the metal interface with the external environment. The estimated corrosion rates were based on values reported in the literature for more aggressive environments that disrupt the protective layer and allow corrosion of the alloy to proceed. In the Hanford subsurface environment, the passive layer would be more likely to remain intact, resulting in lower corrosion rates than those assumed for this analysis.

4.7.2 Hydraulic Impact of a Resource Conservation and Recovery Act Barrier

This analysis did not account for the effect of a Resource Conservation and Recovery Act (RCRA) barrier that may be placed over the metal components in the burial ground, nor was water flow and nickel adsorption simulated in the backfill material that would surround and overlie the metal components. The RCRA barrier would be composed of a geosynthetic membrane and natural, relatively impermeable clay materials, and it would act as an underground umbrella to shield the components from large amounts of infiltrating water. Any water shed by the RCRA barrier would be redirected to soils outside the waste disposal area, thus reducing the amount of infiltration that could

contact the components as long as the barrier remains intact. The clay material used for the barrier is often characterized by its saturated hydraulic conductivity, K_s . The barrier is designed to divert infiltrating water at recharge rates that exceed K_s , and would be expected to transmit water at recharge rates lower than, or equal to, K_s .

Excluding the hydraulic influence of the RCRA barrier in this analysis of water flow and nickel transport is expected to overestimate both the quantity of water contacting the components and the concentration and mass flux of nickel in the water carried away from the burial site during the functional life of the barrier. Even after the geosynthetic membrane fails, the natural barrier material, along with any surface vegetation that may be present, could continue to divert infiltrating water. Therefore, the models employed in this analysis should be conservative for the early post-disposal period, particularly at the 6-cm/yr recharge rate, because the influence of the RCRA barrier would be greater at the higher infiltration rate. Although the barrier has a minimum design life of approximately 30 yr, the processes that would act to degrade or alter the performance of the RCRA barrier occur over very long time periods, and the effective life of the barrier *in situ* should be considerably greater than its minimum design life. However, the performance of the barrier is highly uncertain over the very long time spans required for release of nickel from the components; therefore, the potential hydraulic influence of the RCRA barrier was not included in this analysis.

4.7.3 Recharge Rate with a Protective Barrier

A protective multilayered soil/rock barrier system is also being considered for use in shallow-land burial sites at Hanford. This barrier is of a different design than the RCRA membrane and clay cover discussed in Section 4.7.2, and the protective barrier may be installed either with or without the additional RCRA cover. The protective barrier could potentially eliminate water infiltration over hundreds or even thousands of years (DOE 1987). In addition, it would be designed to minimize wind and water erosion and to provide a deterrent to animal or human intrusion over the long term. However, degradation of the barrier is likely to occur during the time spans

considered in this analysis, and there is substantial uncertainty in predicting its effectiveness over these very long periods.

Analyses conducted to date have shown that a barrier constructed with appropriate materials should not permit any deep recharge at double the rates assumed for this analysis. However, the present estimate of recharge through the barrier (i.e., 0.1 cm/yr) was used for transport modeling because of three considerations: 1) field measurement instruments always introduce measurement error (thus, absolute zero recharge cannot be measured); 2) computer simulations of barrier performance always introduce round-off and truncation error (thus, absolute zero recharge cannot be based solely on computer simulation); and 3) uncertainty in stochastic aspects of both the climate and barrier performance give rise to low-probability scenarios, such as a sequence of high-precipitation years, which may cause water to drain through the barrier system. The high precipitation scenario was hypothesized to result in 5 cm of recharge through the barrier over a 100-yr period, which amounts to an average of 0.05 cm/yr of long-term recharge. This is the performance objective of the barrier system; therefore, use of 0.1 cm/yr as the estimated recharge rate through the protective barrier in the model represents a conservative treatment of this low-probability scenario during the life of the barrier. The 0.5-cm/yr and 6.0-cm/yr recharge rates used to represent long-term climatic conditions at the burial site in the absence of an engineered barrier may also be considered to represent the situation that would occur if a barrier were initially installed, but had degraded over time.

4.7.4 Omission of Lateral or Transverse Dispersion

In reality, lateral dispersion within the vadose zone could result in lower peak concentrations of nickel in solution compared with those predicted by the one-dimensional streamtube model used for this analysis. This would decrease the concentration of nickel in the groundwater at some points in the unconfined aquifer and increase it at other locations. Therefore, the long-term release scenario could result in some vadose zone water reaching the water table at a concentration below the solubility limit. This analysis was conservative in the assumption that all vadose zone water will move vertically

downward and is not mixed laterally with the adjacent clean water, eventually reaching the water table at the estimated solubility concentration for nickel.

Lateral dispersion within the unconfined aquifer could also result in lower peak concentrations of nickel at the 5000-m well and in water discharging to the Columbia River, although it would not affect the total quantity of nickel entering the river annually. Currently, the streamtube model embodied in the TRANSS code neglects all transverse or lateral dispersion. Estimated nickel concentrations at a 100-m well would probably not be reduced significantly by using a multidimensional transport and dispersion model of the unconfined aquifer because of the proximity of the well to the waste site. However, omission of lateral dispersion probably resulted in conservative estimates for nickel concentrations at the more distant downgradient locations.

4.7.5 Extent of Basalt Above the Water Table

Several uncertainties exist in the current model of the unconfined aquifer, making it difficult to estimate the extent of basalt above the water table in the vicinity of the 218-E-12B Burial Ground. The uncertainty in the model for the burial ground is due to the limited information available on water levels and on the geometry of the geologic units in the area, as well as the current lack of spatial resolution in the groundwater flow model.

Currently, the waste water disposal operation at B Pond has a significant influence on the water table in the vicinity of the 218-E-12B Burial Ground. In this area of the site, the unconfined aquifer is located in the lowest sequence of the Hanford formation and is estimated to be less than 3-m thick. A comparison of pre-Hanford water levels (Newcombe et al. 1972) to current water levels (Evans et al. 1988) in the area indicates that the presence of groundwater in the Hanford formation immediately below the burial site may result from liquid discharges to B Pond. When the liquid discharges to B Pond are discontinued as a result of decommissioning 200-East Area facilities, the water table will drop and the basalt formation beneath the burial ground may again extend above the water table. The bottom of the Hanford formation is considered to be the base of the unconfined aquifer in the immediate vicinity of the burial ground because the saturated hydraulic

conductivity of the underlying basalt is several orders of magnitude lower than that of sediments in the Hanford formation. Therefore, the basalt is assumed to be relatively impermeable compared with the sedimentary strata overlying it. The current model was established in the mid-1980s, and only recently have data been available to improve our knowledge of the interface between these formations near the burial ground.

The current model of groundwater flow in the unconfined aquifer, which is based on the CFEST code, is two-dimensional and does not incorporate variations in the vertical dimension. The model is also regional in scale and does not contain sufficient resolution to represent local features in the vicinity of the burial ground. It was designed to reproduce groundwater flow and contaminant transport phenomena to the south and east of the separations facilities located in the 200-East Area, and to the north of facilities located in the 200-West Area. The principal inconsistency with newly available data has been the model's prediction of a significant expanse of basalt above the water table. Additional conservatism is added to the TRANSS transport calculations because of the approach taken to estimate the time required for infiltrating water to move downslope off the basalt surface and drain into the unconfined aquifer. This analysis neglected travel time required for water to traverse the relatively impermeable surface of the bedrock and enter the unconfined aquifer. This approach to estimating a segment of the overall travel time from source to downgradient locations was adopted because of uncertainty about the extent of basalt above the water table.

4.8 SUMMARY

Estimates of nickel migration (as ^{59}Ni and elemental nickel) from the burial ground, and the conservative assumptions incorporated into the modeling, were described in this section of the report. Although ^{63}Ni would initially be present in the components, concentrations of this radionuclide in the unconfined aquifer would be negligible at all times because of the long transport times predicted in this analysis. A peak concentration in the unconfined aquifer of 2.0 pCi/L ^{59}Ni (5.1×10^{-2} mg/L elemental nickel) and a peak annual release of 1.3×10^7 pCi/yr ^{59}Ni (2.2×10^3 g/yr elemental

nickel) to the Columbia River were estimated for the 6.0-cm/yr recharge case. At the 0.5- or 0.1-cm/yr recharge rates, the maximum groundwater concentrations and the annual quantity of nickel reaching the river would be significantly lower. The peak concentrations at all recharge rates were not predicted to occur for tens to hundreds of thousands of years. The results of the nickel transport analyses indicate that a protective infiltration barrier on the surface of the 218-E-12B Burial Ground could have a significant impact on the concentrations that are eventually observed in groundwater, if it had a sufficiently long life span to reduce the infiltration rate for an extended period of time. However, it is unlikely that any engineered barrier would provide protection against water infiltration over the time periods that would be required to completely leach nickel from the components (250,000 to over 10 million years in this analysis). The difficulties in predicting performance of the disposal system over this time result from the absence of long-term data on the degradation mechanisms for engineered structures (human civilization is believed to be less than 10,000 years old), as well as from the probability of changes in climate and in the geologic structure of the disposal site over both the short and long term. The results of the transport model for very long times (greater than 100,000 years) are presented in this document with the provision that they are subject to a high degree of uncertainty because of the potential for geologic change, as discussed in the preceding sections. However, the uncertainty was addressed in this analysis by utilizing reasonably conservative assumptions in the transport analysis to predict future environmental impacts.

5.0 PUBLIC DOSE IMPACTS ASSESSMENT FOR NICKEL RADIOISOTOPES

Estimated concentrations of nickel radioisotopes in the unconfined aquifer, and the quantity of these materials entering the Columbia River, were used to determine the potential public impacts resulting from disposal of components containing ^{59}Ni and ^{63}Ni . The potential radiation dose from nickel isotopes was estimated for maximally exposed individuals (MI) using well water at two locations downgradient from the disposal site, and for an individual who uses water from the Columbia River downstream of the groundwater entry point. A potential collective dose for the downstream population using river water was also determined.

5.1 INDIVIDUAL EXPOSURE SCENARIOS

For this analysis, a family farm scenario was assumed to result in the maximum potential dose to a member of the public based on EIS documents recently completed by the U.S. Department of Energy (DOE 1987; 1989). In this scenario water is withdrawn from the river or from a downgradient well located at the upper surface of the unconfined aquifer. The withdrawal rate was assumed to be 1.8×10^7 L/yr, which is based on irrigating a 2-hectare farm at a rate of $150 \text{ L/m}^2\text{-mo}$ for 6 mo/yr.

The quality of well water as it relates to nickel originating at the disposal site is a function of the fraction of the plume intercepted by the groundwater well, the rate at which soil water containing radioactive nickel enters the aquifer, the pumping rate at the well, and the contaminant concentrations in soil water entering the aquifer. It was assumed that vertical mixing of the contaminant plume was limited to the screened interval of the well (2.5 m for the low-recharge cases and 5.0 m for the high-recharge case). River water quality depended on the river flow rate and on the flux of contaminants reaching the river, which in turn was a function of the total quantity of contaminants at the disposal site and the transport parameters used to evaluate each case.

For the farm scenario, it was assumed that an individual operates a farm that draws water from a well either 100 m or 5000 m downgradient from the

disposal site. This individual was assumed to drink and to irrigate crops with well water, producing 100% of his vegetables and animal products on the farm. The MI was also assumed to live at the site year round, spending half time out of doors. See Appendix B, Table 1, for a complete listing of parameters used to evaluate the dose in this scenario.

Because contaminated groundwater may eventually reach the Columbia River, a scenario was evaluated for the maximally exposed individual who uses the river. This individual spends time swimming, boating, and on the shoreline, and eats fish caught in the river. In addition, this individual drinks water from the river, and all the individual's terrestrial and animal food comes from crops irrigated with river water. See Appendix B, Table 2, for a list of parameters relating to this scenario.

5.2 POPULATION EXPOSURE SCENARIO

Because of the length of time required for nickel to reach the river, the population using water affected by effluents from the disposal site was assumed to be within 80 km of the disposal site, and to remain at an equilibrium level of about 3 million people (about 10 times the current population) over the 10,000 years included in the analysis. The entire population was assumed to drink water from the river and to obtain food from local sources that use river water for irrigation. External exposure to contaminated ground and inhalation of resuspended soil were also considered in the population scenario. Ingestion of fish was not included because the total quantity of fish harvested from the most affected portion of the river (15,000 kg/yr, Woodruff and Hanf 1992) is insufficient to make a significant contribution to the total dose. The dose contribution from irrigated terrestrial foods is several orders of magnitude greater than that from fish ingestion. See Appendix B, Table 3, for a complete listing of parameters related to the population scenario.

5.3 DOSIMETRY AND EXPOSURE ESTIMATES

Radiation doses were based on the standard dosimetry assumptions embodied in ICRP Publications 26, 30, and 48 (1977, 1979-1982, 1986) as

implemented in the GENII computer code (Napier et al. 1988). The code was used to estimate dose to the maximally exposed individual receptors, as well as the collective population dose over 10,000 years for downstream residents on the river. It is assumed that pathways to exposure occur from the use of contaminated ground- or river water for drinking, and for irrigation of terrestrial vegetables and animal fodder. External exposure results from time spent working on irrigated land, and from recreational activities on or near the river. Doses were expressed as the effective dose equivalent (EDE) for this assessment. The EDE includes the annual external dose, and the committed internal dose accumulated over 50 years following a 1-year intake period for ingested or inhaled radionuclides. This is the dose currently used for public dose calculations and for occupational record keeping by DOE and the U.S. Nuclear Regulatory Commission. The committed dose is used as a measure of the potential long-term impact of intakes resulting from accidents and routine releases.

For the farm scenario, the GENII code was executed using a unit radionuclide concentration in groundwater. The resulting dose estimate per unit concentration was then multiplied by the estimated peak concentrations in groundwater for each case. Similarly, the river scenario was evaluated for a unit release of radionuclides to the river, and the result was multiplied by the estimated peak annual flux.

The DITTY (Dose Integrated over Ten Thousand Years) module of the GENII code was used to calculate an integrated collective dose to the downriver population for the peak 10,000-year period in each case. For the integrated dose it was assumed that the population reaches equilibrium at 10 times the current level (DOE 1987). Census data from 1990 for an 80-km zone around the Hanford Site 300 area were used as the baseline for the downriver population estimate.

The entire population was assumed to obtain drinking water from the river and to consume food products irrigated with river water. Because the arrival time of nickel at the water table and the river is tens to hundreds of thousands of years after initial disposal, dose calculations were made for the 10,000-year period beginning at the time of the peak concentration, assuming

that the peak concentration persisted over the entire period. Because nickel concentrations in the river build up rapidly, followed by a more gradual decline, this provides a conservative estimate of population dose integrated over 10,000 years. The actual time period over which this dose accumulates will likewise be several tens of thousands of years in the future, the specific dates depending on transport assumptions for each case. Table 4.6 includes the arrival times of peak flux or concentration for ^{59}Ni .

5.4 RESULTS OF DOSE ANALYSES

All radiation doses resulted from exposure to ^{59}Ni . The first arrival time of contaminated groundwater at an individual receptor location is at least 68,000 years after disposal. Because of the short radiological half-life of ^{63}Ni (100 yr) relative to transport time, the entire inventory of ^{63}Ni would decay to negligible levels before it could reach the water table. Only ^{59}Ni , with a longer half-life of 76,000 yr, would be present in significant quantities over the time scale of this analysis.

Results of the individual and collective dose calculations are shown in Table 5.1, and listings of the parameters used in the calculations are presented in Appendix B, Tables 1 through 3. The annual dose to a maximally exposed individual in the family farm scenario for a 1 pCi/L concentration of ^{59}Ni in well water would be 4.8×10^{-4} mrem; the corresponding dose to an individual for release of 1 pCi/y to the Columbia River would be 1.4×10^{-17} mrem. The dose estimates for unit groundwater concentrations and surface water releases were multiplied by the peak concentrations or releases presented in Table 4.5 to obtain the dose estimates in Table 5.1.

The potential annual dose to the MI using groundwater from a well located 100 m from the full-array disposal site is less than 0.00097 mrem/yr from all pathways. Of this total, the drinking water pathway accounts for less than 0.0003 mrem/yr at the maximum concentration. The dose for the MI using a well 5000 m from the disposal site was less than 0.00054 mrem/yr, and an individual using river water would receive an insignificant dose. The collective population dose over 10,000 years is less than 0.00053 person-rem. The high end of the dose range represents assumed release conditions for a

TABLE 5.1. Peak Dose to the Maximally Exposed Individual from ^{59}Ni in Downgradient Groundwater Wells and Dose to an Individual or Collective Population Using the Columbia River as a Function of Release Conditions from the 218-E-12B Burial Ground

Receptor Location	Recharge Rate	Peak Dose to a Maximally Exposed Individual (mrem/yr)	
		Single Component	Multiple Components (4 X 30 Array)
100-m Well	0.1 cm/yr	4.8E-19	2.8E-18
	0.5 cm/yr	5.9E-07	3.3E-06
	6.0 cm/yr	1.7E-04	9.7E-04
5000-m Well	0.1 cm/yr	9.1E-20	8.6E-19
	0.5 cm/yr	1.7E-07	1.4E-06
	6.0 cm/yr	7.4E-05	5.4E-04
River	0.1 cm/yr	<1.0E-20	<1.0E-20
	0.5 cm/yr	2.6E-16	1.2E-13
	6.0 cm/yr	6.0E-13	1.8E-10
10,000-Year Collective Dose to Columbia River Population (person-rem)			
River	0.1 cm/yr	<1.0E-20	9.4E-20
	0.5 cm/yr	8.0E-10	3.8E-07
	6.0 cm/yr	1.9E-06	5.3E-04

more humid climate than currently exists, a high groundwater recharge rate, and an array of 120 components in the disposal site. This combination represents the most conservative set of assumptions used in the analysis and provides a high-end estimate of the dose to an individual or population. The low end of each dose range is based on an arid climate similar to existing conditions with a lower groundwater recharge rate. These assumptions reflect the more probable set of conditions based on experimental data presented in the previous sections.

The potential dose received by the MI via the family farm scenario in the future can be compared to the dose received by an individual in the local area from natural background radiation today. The MI may receive an annual dose of up to 0.001 mrem at a time 68,000 years after disposal. For perspective, exposure to natural background radiation today results in an annual per capita effective dose equivalent of about 300 mrem/yr, with naturally occurring radon and its radioactive decay products accounting for two-thirds of the total (about 200 mrem/yr average in the United States). Anthropogenic (man-made) sources contribute another 65 mrem/yr, of which medical exposure accounts for approximately 53 mrem/yr averaged over the U.S. population (NCRP 1987).

The maximum collective dose to the downriver population over 10,000 years would be 0.00053 person-rem assuming that the population had reached an equilibrium level of about 3 million residents using the river. The estimated collective dose from disposal site effluents is many orders of magnitude lower than the collective dose to 3 million people from natural background radiation, which amounts to approximately 9 billion person-rem over 10,000 years (based on an average individual dose of 300 mrem/yr). The collective dose from ⁵⁹Ni is also substantially lower than the dose that would be expected to result in a single cancer fatality (2000 person-rem) over this 10,000-year period.

5.5 SUMMARY OF DOSE ASSESSMENT

For this analysis, concentrations of radioactive nickel in the unconfined aquifer and the Columbia River were evaluated using standard dosimetric

methods. Even for the most conservative set of assumptions in the analysis, predicted levels of radionuclides would not be expected to exceed current applicable state or federal guidelines.

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APPENDIX A

EXPERIMENTAL DATA FOR BATCH ADSORPTION STUDIES

	A	B	C	D	E	F	G	H	I	J
1	Sample Id	Wt. Sed	Solution Left	Solution Weigh	Conc. NI cold	7D NI conc. C	NI uCl/ml	10D NI uCl/ml	Kd C N° 7D	Kd R NI10D
2		grams	after wash	grams	(Inlt.) ppb	ppb	(Inlt.)			(Inlt.)
3	E7A-7	0.989	0.594	37.442			1.430E-04	5.425E-05		6.135E+01
4	E7B-7	1.035	0.590	38.222			1.436E-04	6.204E-05		4.796E+01
5	E8A-7	1.011	0.613	38.100			1.416E-04	4.383E-05		8.348E+01
6	E8B-7	1.031	0.511	38.834			1.414E-04	4.020E-05		9.427E+01
7	E9A-7	0.984	0.546	38.831			1.432E-04	2.243E-05		2.119E+02
8	E9B-7	1.114	0.695	39.525			1.401E-04	2.154E-05		1.946E+02
9	E10A-7	0.999	0.572	38.692			1.415E-04	1.457E-05		3.369E+02
10	E10B-7	0.988	0.942	38.800			1.494E-04	1.990E-05		2.546E+02
11	E11A-7	0.993	0.596	37.359	2550	440.5			1.796E+02	
12	E11B-7	0.960	1.453	35.936	2470	477			1.549E+02	
13	E12A-7	0.968	0.590	38.767	512	131			1.159E+02	
14	E12B-7	1.001	0.541	37.619	491	111			1.281E+02	
15	E17A-7	1.009	0.686	38.993			1.470E-04	6.066E-05		5.434E+01
16	E17B-7	1.005	0.407	39.332			1.439E-04	5.486E-05		6.312E+01
17	E17C-7	1.007	0.554	38.982			1.455E-04	6.137E-05		5.249E+01
18	E18A-7	0.977	0.528	39.783			1.500E-04	5.301E-05		7.400E+01
19	E18B-7	1.015	0.516	39.732			1.501E-04	4.336E-05		9.590E+01
20	E18C-7	1.018	0.446	40.499			1.501E-04	4.693E-05		8.702E+01
21	E19A-7	0.964	0.435	40.192			1.458E-04	2.882E-05		1.687E+02
22	E19B-7	0.995	0.412	40.232			1.478E-04	3.247E-05		1.433E+02
23	E19C-7	0.996	0.530	39.722			1.468E-04	2.816E-05		1.675E+02
24	E20A-7	1.004	1.658	38.506			1.418E-04	1.550E-05		3.109E+02
25	E20B-7	1.016	0.481	39.797			1.421E-04	1.864E-05		2.590E+02
26	E20C-7	1.005	0.528	40.254			1.419E-04	1.974E-05		2.474E+02
27	E21A-7	1.005	0.810	37.799	2380	464			1.545E+02	
28	E21B-7	0.994	0.595	38.227	2520	586			1.263E+02	
29	E22A-7	0.997	1.016	37.973	2570	466.5			1.707E+02	
30	E22B-7	1.017	0.907	38.243	2500	488			1.541E+02	
31	E23A-7	1.018	0.407	39.305	2330	310			2.512E+02	
32	E23B-7	0.978	0.551	39.346	2550	499			1.648E+02	
33	E24A-7	1.000	0.435	38.399	2540	429			1.885E+02	
34	E24B-7	0.993	0.490	36.732	2530	404			1.942E+02	
35	E25A-7	1.051	0.397	39.271	2540	423			1.866E+02	
36	E25B-7	1.020	0.388	38.826	2520	416.5			1.919E+02	
37	E26A-7	0.999	0.607	37.220	526	126.5			1.171E+02	
38	E26B-7	1.023	0.454	38.154	497	113			1.263E+02	
39	E27A-7	1.026	0.622	38.299	97.9	17.7			1.685E+02	
40	E27B-7	1.006	0.755	37.651	96	17.9			1.625E+02	
41	E28A-7	1.006	0.527	39.636	19.5	4.51			1.304E+02	
42	E28B-7	1.009	0.634	37.653	22.5	5.39			1.178E+02	
43										

[illegible]

APPENDIX B

INPUT PARAMETERS AND RESULTS OF RADIOLOGICAL DOSE CALCULATIONS
FOR INDIVIDUALS AND POPULATIONS USING THE GENII COMPUTER CODE

**TABLE B.1. Unit Dose Calculation Parameters for an Individual
Using Well Water**

Parameter Description	Parameter Value or Setting
Input Data Files and Software Libraries	
Input file name: FARMUNIT.in 2-14-92	
GENII Default Parameter Values (28-Mar-90 RAP)	
Radionuclide Master Library (11-28-90 RAP)	
Food Transfer Factor Library (RAP 29-Aug-88)	
Bioaccumulation Factor Library (30-Aug-88) RAP	
External Dose Factors for GENII ... (8-May-90 RAP)	
Internal Dose Increments, PNL Solubility Choices Rerun (12-3-90 PDR)	
Scenario Description	
Scenario Type:	Near Field
Release:	Chronic
Dose to:	Individual
Exposure Pathways:	Ground, external Inhalation uptake Drinking water ingestion Terrestrial foods ingest. Animal product ingestion
Exposure and Dose Calculation Times	
Intake Period:	1 year
Dose Commitment Period:	50 years

TABLE B.1. (cont.)

Parameter Description	Parameter Value or Setting
Basic Concentrations in Environmental Media	Groundwater
Ni-59	1.0 pCi/L
Near Field Parameters	
Fraction of roots in upper soil	1.0
External Exposure	
Hours of exposure to ground contamination	4.4E+03
Residential irrigation source	1
1-ground water	
2-surface water	
Residential irrigation application rate (in./yr)	4.0E+01
Residential irrigation duration (mo/yr)	6.0E+00
Inhalation Exposure	
Hours of inhalation exposure per year	8.8E+03
Resuspension model:	1
1-Mass Loading	
2-Anspaugh	
Mass loading factor (g/m ³)	1.0E-04
Drinking Water Source and Irrigation	
Drinking water consumption rate (l/yr)	7.3E+02
Drinking water source:	1
1-ground	
2-surface	
3-system	
Drinking water treatment: T/F	F
Drinking water transit/holdup time (d)	0.0

TABLE B.1. (cont.)

Parameter Description		Parameter Value or Setting			
Terrestrial Food Production		--IRRIGATION--		GROW	
		S RATE	TIME	TIME	YIELD
	Food Type:	* in./yr	mo/yr	d	kg/m²
		-----	-----	-----	-----
	Leafy Vegetables	1 35.0	6.0	90.0	1.5
	Other Vegetables	1 40.0	6.0	90.0	4.0
	Fruit	1 35.0	6.0	90.0	2.0
	Cereal	1 35.0	6.0	90.0	0.8
Terrestrial Food Ingestion		-CONSUMPTION-			
		HOLDUP	RATE		
	Food Type:	d	kg/yr		
		-----	-----		
	Leafy Vegetables	1.0	3.0E+01		
	Other Vegetables	5.0	1.8E+02		
	Fruit	5.0	3.3E+02		
	Cereal	5.0	8.0E+01		
		* Irrigation Source:			
		1-groundwater			
		2-surface water			
Animal Food Production		-----STORED FEED-----			
		-IRRIGATION--		GROW	
		S RATE	TIME	TIME	YIELD
	Food Type:	* in./yr	mo/yr	d	kg/m³
		-----	-----	-----	-----
	Meat	1 35.0	6.0	90.0	0.80
	Poultry	1 0.0	0.0	90.0	0.80
	Milk (Cow)	1 47.0	6.0	30.0	2.00
	Eggs	1 0.0	0.0	90.0	0.80
		-STORED FEED-		DRINKING	
		DIET	STOR-	WATER	
		FRAC-	AGE	CONTAM.	
		TION	d	FRACTION	
		-----	-----	-----	
	Meat	0.25	180.0	1.00	
	Poultry	1.00	180.0	1.00	
	Milk (Cow)	0.25	100.0	1.00	
	Eggs	1.00	180.0	1.00	

TABLE B.1. (cont.)

Parameter Description	Parameter Value or Setting			
Animal Food Production (cont.)	-----FRESH FORAGE-----			
	--IRRIGATION--		GROW	
	S RATE	TIME	TIME	YIELD
	* in./yr	mo/yr	d	kg/m ³

Meat	1	47.0	6.0	45.0
Milk (Cow)	1	47.0	6.0	30.0

	-FRESH FORAGE-			
	DIET	STOR-		
	FRAC-	AGE		
	TION	d		

Meat	0.75	100.0		
Milk (Cow)	0.75	0.0		

Animal Food Ingestion	-----HUMAN-----			
	CONSUMPTION			
	RATE	HOLDUP		
	kg/yr	d		

Food Type:				
Meat	8.0E+01	15.0		
Poultry	1.8E+01	1.0		
Milk (Cow)	2.7E+02	1.0		
Eggs	3.0E+01	1.0		

	* Irrigation Source:			
	1-groundwater			
	2-surface water			

TABLE B.2. Unit Dose Calculation Parameters for an Individual Using Water from the Columbia River

Parameter Description	Parameter Value or Setting
Input Data Files and Software Libraries	
Input file name: RIVRUNIT.in	2-14-92
GENII Default Parameter Values	(28-Mar-90 RAP)
Radionuclide Master Library	(11-28-90 RAP)
Food Transfer Factor Library	- (RAP 29-Aug-88)
Bioaccumulation Factor Library	(30-Aug-88) RAP
External Dose Factors for GENII ...	(8-May-90 RAP)
Internal Dose Increments, PNL Solubility Choices Rerun	(12-3-90 PDR)
Scenario Description	
Scenario Type:	Far Field
Release:	Chronic
Dose to:	Individual
Transport Mode:	Surface Water
Exposure Pathways:	Ground, external Recreation, external Inhalation uptake Drinking water ingestion Aquatic foods ingestion Terrestrial foods ingest. Animal product ingestion
Exposure and Dose Calculation Times	
Release Period:	1 year
Intake Period:	1 year
Dose Commitment Period:	50 years

TABLE B.2. (cont.)

Parameter Description	Parameter Value or Setting
Release Terms	Surface water
Ni-59	1.0 pCi/yr
Surface Water Transport	
Mixing ratio model:	0
0-use value	
1-river	
2-lake	
3-river flow	
Transit time to irrigation withdrawal (h)	0.0E+00
Average river flow rate (m ³ /sec)	3.0E+03
Mixing ratio (dimensionless)	1.0E+00
External Exposure	
Hours of exposure to ground contamination	4.4E+03
Residential irrigation source:	2
1-ground water	
2-surface water	
Residential irrigation application rate (in./yr)	4.0E+01
Residential irrigation duration (mo/yr)	6.0E+00
Hours of exposure from swimming	1.0E+02
Hours of exposure from boating	1.0E+02
Hours of exposure from shoreline activities	5.0E+02
Shoreline type:	1
1-river	
2-lake	
3-ocean	
4-tidal basin	
Surface water transit time to recreational site (h)	8.0E+00
Inhalation Exposure	
Hours of inhalation exposure per year	8.8E+03
Resuspension model:	1
1-Mass Loading	
2-Anspagh	
Mass loading factor (g/m ³)	1.0E-04

TABLE B.2. (cont.)

Parameter Description	Parameter Value or Setting		
Drinking Water Source and Irrigation			
Drinking water consumption rate (L/yr)	7.3E+02		
Drinking water source:	2		
1-ground			
2-surface			
3-system			
Drinking water treatment:	F		
T = Yes			
F = No			
Drinking water transit/holdup time (d)	0.0		
Aquatic Food Ingestion			
Food Type:	-CONSUMPTION- HOLDUP RATE	TRANSIT TIME	
	d kg/yr	h	
	----	-----	
Fish	1.0 4.0E+01	1.0	
Terrestrial Food Production			
Food Type:	--IRRIGATION-- S RATE TIME	GROW TIME YIELD	
	* in./yr mo/yr	d kg/m ²	
	----	-----	
Leafy Vegetables	2 35.0 6.0	90.0	1.5
Other Vegetables	2 40.0 6.0	90.0	4.0
Fruit	2 35.0 6.0	90.0	2.0
Cereal	2 35.0 6.0	90.0	0.8
Terrestrial Food Ingestion			
Food Type:	-CONSUMPTION- HOLDUP RATE		
	d kg/yr		

Leafy Vegetables	1.0 3.0E+01		
Other Vegetables	5.0 1.8E+02		
Fruit	5.0 3.3E+02		
Cereal	5.0 8.0E+01		

TABLE B.2. (cont.)

Parameter Description		Parameter Value or Setting				
Animal Food Production		-----STORED FEED-----				
		-IRRIGATION--		GROW		
		S RATE	TIME	TIME	YIELD	
		* in./yr	mo/yr	d	kg/m ³	
		-----		-----		
	Food Type:					
	Meat	2	35.0	6.0	90.0 0.80	
	Poultry	2	35.0	6.0	90.0 0.80	
	Milk (Cow)	2	47.0	6.0	30.0 2.00	
	Eggs	2	35.0	6.0	90.0 0.80	
		-----		-----		
		-STORED FEED-		DRINKING		
		DIET	STOR-	WATER		
		FRAC-	AGE	CONTAM.		
		TION	d	FRACTION		
		----		-----		
	Meat	0.25	180.0	1.00		
	Poultry	1.00	180.0	1.00		
	Milk (Cow)	0.25	100.0	1.00		
	Eggs	1.00	180.0	1.00		
		-----		-----		
		-----FRESH FORAGE-----		-----		
		--IRRIGATION--		GROW		
		S RATE	TIME	TIME	YIELD	
		* in./yr	mo/yr	d	kg/m ³	
		-----		-----		
	Meat	2	47.0	6.0	45.0 2.00	
	Milk (Cow)	2	47.0	6.0	30.0 1.50	
		-----		-----		
		-FRESH FORAGE-				
		DIET	STOR-			
		FRAC-	AGE			
		TION	d			
		----		-----		
	Meat	0.75	100.0			
	Milk (Cow)	0.75	0.0			

TABLE B.2. (cont.)

Parameter Description		Parameter Value or Setting	
Animal Food Ingestion		-----HUMAN-----	
		CONSUMPTION	
		RATE	HOLDUP
Food Type:		kg/yr	d
		-----	-----
Meat		8.0E+01	15.0
Poultry		1.8E+01	1.0
Milk (Cow)		2.7E+02	1.0
Eggs		3.0E+01	1.0
		* Irrigation Source:	
		1-groundwater	
		2-surface water	

TABLE B.3. Dose Calculation Parameters for Collective Population Using Water from the Columbia River

Parameter Description	Parameter Value or Setting
Input Data Files and Software Libraries	
Input file name: DITTY.in	11-22-92
Radionuclide Master Library	(11-28-90 RAP)
Food Transfer Factor Library	- (RAP 29-Aug-88)
Bioaccumulation Factor Library	- (30-Aug-88) RAP
External Dose Factors for GENII...	(8-May-90 RAP)
DITTY Internal Dose Factors, PNL Solubility Choices Rerun 12-3-90 PDR	
Scenario Description	
Scenario Type:	Far Field
Release:	Chronic
Dose to:	Population
Transport Mode:	Surface Water
Exposure Pathways:	Ground, external Recreation, external Inhalation uptake Drinking water ingestion Aquatic foods ingestion Terrestrial foods ingest. Animal product ingestion
Exposure and Dose Calculation Times	
Release Period:	10,000 years
Intake Period:	10,000 years
Dose Commitment Period:	50 years
Release Terms	
Ni-59	Surface water Case-specific release rates from groundwater transport model (pCi/y)

TABLE B.3. (cont.)

Parameter Description	Parameter Value or Setting
Surface Water Transport	
Average river flow rate (ft ³ /sec)	1.5E+02
Mixing ratio (dimensionless)	1.0E+00
Reconcentration ratio (dimensionless)	1.0E+00
External Exposure	
Hours of exposure to ground contamination	4.4E+03
Irrigation rate (L/m ² -mo)	1.5E+02
Irrigation duration (mo/yr)	6.0E+00
Hours of exposure from swimming	1.0E+01
Hours of exposure from shoreline activities	1.7E+01
Drinking Water and Aquatic Food Ingestion	
Drinking water consumption rate (L/yr)	7.3E+02
Fish consumption (kg/yr)	4.0E-01
Terrestrial and Animal Food Ingestion	
	<div> <div>GROWING</div> <div>TIME</div> <div>d</div> <div>-----</div> </div> <div> <div>YIELD</div> <div>kg/m²</div> <div>-----</div> </div> <div> <div>CONSUMPTION</div> <div>RATE</div> <div>kg/yr</div> <div>-----</div> </div>
Food Type:	
Leafy Vegetables	90.0 1.5 1.5E+01
Other Vegetables	90.0 4.0 2.8E+02
Eggs	90.0 0.8 2.0E+01
Milk (Cow)	30.0 1.3 2.3E+02
Beef	90.0 0.8 4.0E+01
Pork	90.0 0.8 4.0E+01
Poultry	90.0 0.8 8.5E+00
Irrigation Source:	2
1-groundwater	
2-surface water	

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