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Hanford Grout Technology Program

Long-Term Performance Assessment of Grouted Phosphate/Sulfate Waste From N Reactor Operations

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April 1987

Prepared for the U.S. Department of Energy
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Pacific Northwest Laboratory
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LONG-TERM PERFORMANCE ASSESSMENT OF GROUTED PHOSPHATE/SULFATE WASTE FROM N REACTOR OPERATIONS

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ABSTRACT

The long-term performance of the grout disposal system for Phosphate/ Sulfate Waste (PSW) was analyzed. PSW is a low-level liquid generated by activities associated with N Reactor operations. The waste will be mixed with dry solids and permanently disposed of as a cementitious grout in sub-surface concrete vaults at Hanford's 200-East Area. Two categories of scenarios were analyzed that could cause humans to be exposed to radionuclides and chemicals from the grouted waste: contaminated groundwater and direct intrusion. In the groundwater scenario, contaminants are released from the buried grout monoliths, then eventually transported via the groundwater to the Columbia River. As modeled, the contaminants are assumed to leach out of the monoliths at a constant rate over a 10,000-year period. The other category of exposure involves intruders who inadvertently contact the waste directly, either by drilling, excavating, or gardening.

Long-term impacts that could result from disposal of PSW grout were expressed in terms of incremental increases of 1) chemical concentrations in the groundwater and surface waters, and 2) radiation doses. None of the calculated impacts exceeded the corresponding regulatory limits set by Washington State, the Department of Energy, or the Nuclear Regulatory Commission.

SUMMARY

The long-term performance of the grout disposal system for Phosphate/Sulfate Waste (PSW) was analyzed. This report documents the performance assessment, which served as input to the Environmental Assessment for the Grouting and Near-Surface Disposal of Low-Level Radioactive Phosphate/Sulfate Waste from N Reactor Operations (U.S. DOE 1986a). The long-term performance of the PSW grout disposal system was evaluated in terms of 1) potential incremental increases of regulated *chemicals* in groundwater and in the Columbia River, and 2) potential incremental *radiation doses* to a person who either intrudes directly into a grout monolith or uses groundwater contaminated by long-term release of PSW radionuclides.

PSW GROUT DISPOSAL SYSTEM

The Phosphate/Sulfate Waste discussed in this report is a low-level radioactive waste generated by activities associated with operation of the N Reactor on the Hanford Site near Richland, Washington. Contaminants in the waste include radionuclides, and nonradioactive chemicals that are regulated by federal and state drinking water standards. The waste will be mixed with grout-forming materials (cement, fly ash, and clays) and then pumped as a slurry into sub-surface concrete vaults on the Hanford Site for hardening and permanent disposal.

The PSW grout disposal site is located in the 200-East area of the Hanford Site. The currently projected volume of PSW will require the use of approximately 12 rectangular vaults made of reinforced concrete. Each vault will be 10.4 m deep, 15.24 m wide, 38.1 m long, and lined on the inside with high-density polyethylene. Each vault will be covered with concrete cover blocks and soil backfill, then filled with radioactive grout up to about 90% of the vault's volume. The remaining 10% of each vault will be filled up to the concrete cover with nonradioactive grout. Finally, an overburden of soil and rock will be placed over the vaults.

MODEL OF GROUT DISPOSAL SYSTEM AND APPROACH OF PERFORMANCE ASSESSMENT

In this performance assessment, we used environmental pathway models to project transport of contaminants from the disposed waste form to points of human access. The transport may be the result of natural release mechanisms (e.g., diffusion) or a consequence of inadvertent disruptive intrusion.

Over very long periods of time (hundreds or thousands of years), the vault and liners are expected to degrade and allow some degree of contaminant migration into the soil layers around and below the vault (vadose zone). Water percolating down through the vadose zone could carry contaminants down to the groundwater and subsequently to uncontrolled areas such as the Columbia River, where humans could be exposed to the contaminants. In addition, because institutional control of the Hanford Site cannot be ensured indefinitely, an inadvertent intruder could conceivably move onto the grout disposal site and be exposed to the grout contaminants by drilling, excavating, or gardening.

The physical transport processes that could result in contamination of the groundwater were modeled using a number of simplifying assumptions. An intact grout monolith was assumed to be in direct contact with the surrounding soil; no isolation credit was taken for the liners, vault, water-shedding cap, or barrier. In this model, the monolith was assumed to be bathed with incoming water and all the contaminants leached out of the monolith at a constant rate over a 10,000-year period. (Laboratory leach tests on small cylinders of simulated grout, in conjunction with the assumption of a diffusional release mechanism and a subsequent release model that can be scaled up to describe field-scale release, indicate that total release from an uncracked monolith would actually occur over a much longer period than 10,000 years; the constant-release model used here is more conservative than the assumed actual release model would be.) Once contaminants enter the vadose zone, they were modeled as traveling to the groundwater via vertical, one-dimensional transport. Some radionuclides travel through the vadose zone more slowly than water, because they are sorbed on the soil. Some of the nonradiological contaminants form

precipitates in the soil, and hence their transport in the vadose zone is solubility-limited. The remainder of the contaminants were modeled as having the same travel time as water.

The amount of water available to carry contaminants through the vadose zone is a fraction of the amount of precipitation. Therefore, the analysis was performed for two recharge rates: one represents an estimate of current climatic conditions, and the other encompasses the effects of a hypothetical wetter climate.

The second means by which humans could become exposed to contaminants from the grout site is by direct intrusion. Active controls of the grout disposal site were assumed to cease 100 years after disposal (although this situation is extremely unlikely). A suite of inadvertent-intruder scenarios was postulated (drilling, excavating, and gardening), and the resulting radiological doses were calculated.

Calculations of radiological dose to humans are directly related to a number of parameters specific to each radionuclide: sorption in the vadose zone, rate of radioactive decay, and effect of the radionuclide on various organs in the human body. Two computer codes were used to calculate the doses associated with various exposure scenarios: the ONSITE/MAXI1 code (Napier et al. 1984) and the DITTY code (Napier, Peloquin and Strenge 1986).

RESULTS

Long-term impacts that could result from disposal of PSW grout are expressed in terms of incremental increases of 1) chemical concentrations in the groundwater and surface waters, and 2) radiation doses. The impacts are then compared to correspondent regulations.

Nonradiological Chemical Impacts

Projected incremental increases in concentrations of regulated chemicals are listed in Table S.1. The concentrations were calculated for two locations: at a hypothetical well 5 km downgradient from the PSW grout disposal site, and in the Columbia River. Although the groundwater below the Hanford

TABLE S.1. Calculated Increase in Concentrations of Regulated Chemicals in a Hypothetical 5-km Well and in the Columbia River, mg/L

	In 5-km Well Water		In Columbia River Water		Washington State Drinking Water Limit
	Recharge Rate, cm/yr				
	0.5	5.0	0.5	5.0	
Primary Contaminants					
Arsenic	2×10^{-2}	1×10^{-2}	7×10^{-10}	7×10^{-10}	0.05
Barium	6×10^{-4}	4×10^{-3}	3×10^{-11}	2×10^{-10}	1.0
Cadmium	3×10^{-5}	2×10^{-4}	1×10^{-12}	1×10^{-11}	0.01
Chromium	1×10^{-3}	1×10^{-2}	4×10^{-11}	6×10^{-10}	0.05
Fluoride	3×10^{-2}	2×10^{-1}	1×10^{-9}	1×10^{-8}	2.0
Lead	4×10^{-3}	3×10^{-2}	2×10^{-10}	2×10^{-9}	0.05
Mercury	1×10^{-4}	9×10^{-5}	5×10^{-12}	5×10^{-12}	0.002
Nitrogen	3×10^{-1}	3×10^{-1}	1×10^{-8}	1×10^{-8}	10.0
Selenium	1×10^{-3}	8×10^{-4}	5×10^{-11}	5×10^{-12}	0.01
Silver	1×10^{-3}	8×10^{-3}	4×10^{-11}	5×10^{-10}	0.05
Secondary Contaminants					
Chloride	3×10^{-1}	3×10^{-1}	2×10^{-8}	2×10^{-8}	250
Copper	8×10^{-4}	7×10^{-3}	4×10^{-11}	4×10^{-10}	1.0
Iron	3×10^{-4}	3×10^{-3}	1×10^{-11}	2×10^{-10}	0.3
Manganese	3×10^{-4}	2×10^{-3}	1×10^{-11}	1×10^{-10}	0.05
Sulfate	3×10^{1}	2×10^{1}	1×10^{-6}	1×10^{-6}	250
Zinc	3×10^{-4}	2×10^{-3}	1×10^{-11}	1×10^{-10}	5.0

Site does not constitute a public water supply, Washington State limits (as given in the Washington Administrative Code) are listed for comparison (WAC 1985). All calculated concentrations are below these limits established for drinking water.

The Hanford Reach of the Columbia River is governed by Class A water quality standards for the State of Washington (WAC 1984). These standards do not list specific concentration limits for inorganic chemicals, but they do include limits on biological waste, turbidity, thermal waste (heat), and aesthetic qualities of the river. The incremental increases of chemicals in

Columbia River water from disposal of PSW grout are projected to be very low and will comply with all Class A water quality standards.

Radiological Impacts

A summary of calculated radiological impacts is provided in Table S.2; these values represent incremental increases in dose. Correspondent regulatory limits are also listed for comparison. Maximum dose to an intruder is compared to limits established in DOE Order 5480.1A (U.S. DOE 1981a) and by the Nuclear Regulatory Commission for shallow-land disposal of commercial low-level wastes (U.S. NRC 1982a). Maximum dose from the drinking water scenario is compared to Washington State limits for a public water supply (WAC 1985). In the short term, strontium-90 and cesium-137 dominate the intruder impacts. The long-term radiological impacts result primarily from uranium-238.

TABLE S.2. Summary of Maximum Radiological Impacts

Scenario	Dose, mrem/yr Total Body/ Critical Organ	Regulatory Dose Limit, mrem/yr Total Body/ Critical Organ	Dominant Radionuclide
Drinking Water, 0.5 cm/yr recharge	0.02/0.3	4/4[a]	^{238}U
Full Garden, 5.0 cm/yr recharge	0.05/0.4	25/75[b]	^{238}U
River, both recharges	$4 \times 10^{-9}/3 \times 10^{-8}$	25/75[b]	^{238}U
Intruder (residential home garden)	60/200	500/1500[b,c]	^{90}Sr, ^{137}Cs

(a) WAC (1985).
(b) U.S. NRC (1982a).
(c) U.S. DOE (1981a).

PURPOSE AND CONCLUSIONS OF PERFORMANCE ASSESSMENT

This performance assessment was prepared as input to the environmental assessment for PSW grout. An environmental assessment "means a concise public document...that serves to briefly provide sufficient evidence and analysis for determining whether to prepare an environmental impact statement or a finding of no significant impact" (U.S. EPA 1985a). Thus, an environmental assessment is a scoping document written to assist decisionmakers.

Environmental assessments, and other such documents described by the National Environmental Policy Act (NEPA) of 1969, typically include a "bounding analysis" to help foster what NEPA calls "excellent decisions." A bounding analysis is performed with a set of data, modeling assumptions, and accidental release scenarios, the total of which is sufficiently conservative so that there is a high degree of confidence that as a result of the compounded conservatisms, the calculated (predicted) environmental impacts will exceed those expected in practice. Individual parameters are not necessarily extreme values. They may be mean values in cases where ample data warrants such a choice, or values well on the conservative side of the expected mean for parameters with highly uncertain ranges of values. Furthermore, when uncertainties exist regarding modeling assumptions, the assumptions are made on what is considered to be the side of conservatism. Finally, accident scenarios are chosen, within the bounds of credibility, to describe the most serious incidents (i.e., those with the greatest impacts) that could reasonably occur. (For this performance assessment, "accident scenarios" were the scenarios of inadvertent intrusion into the grout disposal site.)

To support the environmental assessment for grout disposal of PSW, the authors of this long-term performance assessment used models and the best available data to provide what is believed to be a "bounding analysis" such as that described above. The results can then be used as input for determining whether to prepare an environmental impact statement or a finding of no significant impact.

To assist decisionmakers, calculated impacts were compared to potentially applicable regulations. It is not precisely known what regulations will apply to the Hanford Site and surrounding areas during the post-operational period of disposal. Regulatory requirements were conservatively assumed to apply for a period of 10,000 years following disposal. None of the calculated impacts exceeded the regulatory limits that were reviewed and provided for comparison. Therefore, the results of this performance assessment indicate that grout disposal of PSW can provide long-term protection of public health and safety.

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FIGURES

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1.0 INTRODUCTION

This report documents the long-term performance assessment that is described in DOE/EA-0312, Environmental Assessment for the Grouting and Near-Surface Disposal of Phosphate/Sulfate Waste from N Reactor Operations (U.S. DOE 1986a).

The Environmental Assessment (EA) was prepared for the Grout Disposal Program, which is managed by Rockwell Hanford Operations (Rockwell). The EA applies specifically to Phosphate/Sulfate Waste from N Reactor Operations (PSW), a low-level radioactive waste stream scheduled for grout disposal. In this application, grout is a mixture of liquid wastes and grout-forming solids (portland cement, fly ash, and various clays) that will be pumped as a slurry to sub-surface reinforced concrete vaults where it will subsequently harden into a solid matrix that immobilizes the waste. The primary purpose of the performance assessment (PA) contained in the EA was to investigate whether the grout disposal system for PSW grout could provide long-term protection of public health and safety. To accomplish this goal, Pacific Northwest Laboratory (PNL) researchers investigated the pathways and mechanisms by which wastes could conceivably be transported to the biosphere. The results of the study were then compared with potentially applicable federal and state regulatory requirements.

This report begins with a discussion on the rudiments of a PA (Chapter 2.0). The progression and organization of the remainder of the report are shown in Figure 1.1. Chapter 3.0 describes the PSW waste stream and the grout disposal system. Chapter 4.0 describes the inventories of contaminants (radionuclides and inorganic chemicals as regulated by drinking water standards) in the PSW grout and how we model the release of contaminants from grout. Chapter 5.0 describes the simulated transport of contaminants in the soil surrounding the PSW grout disposal site and the subsequent movement of the contaminants in Hanford groundwater to points of access by humans. Chapter 6.0 describes the method of calculating radiological doses projected to result from hypothetical transport of radionuclides to the biosphere. For the reader's convenience, technical terms are defined in Chapter 7.0.

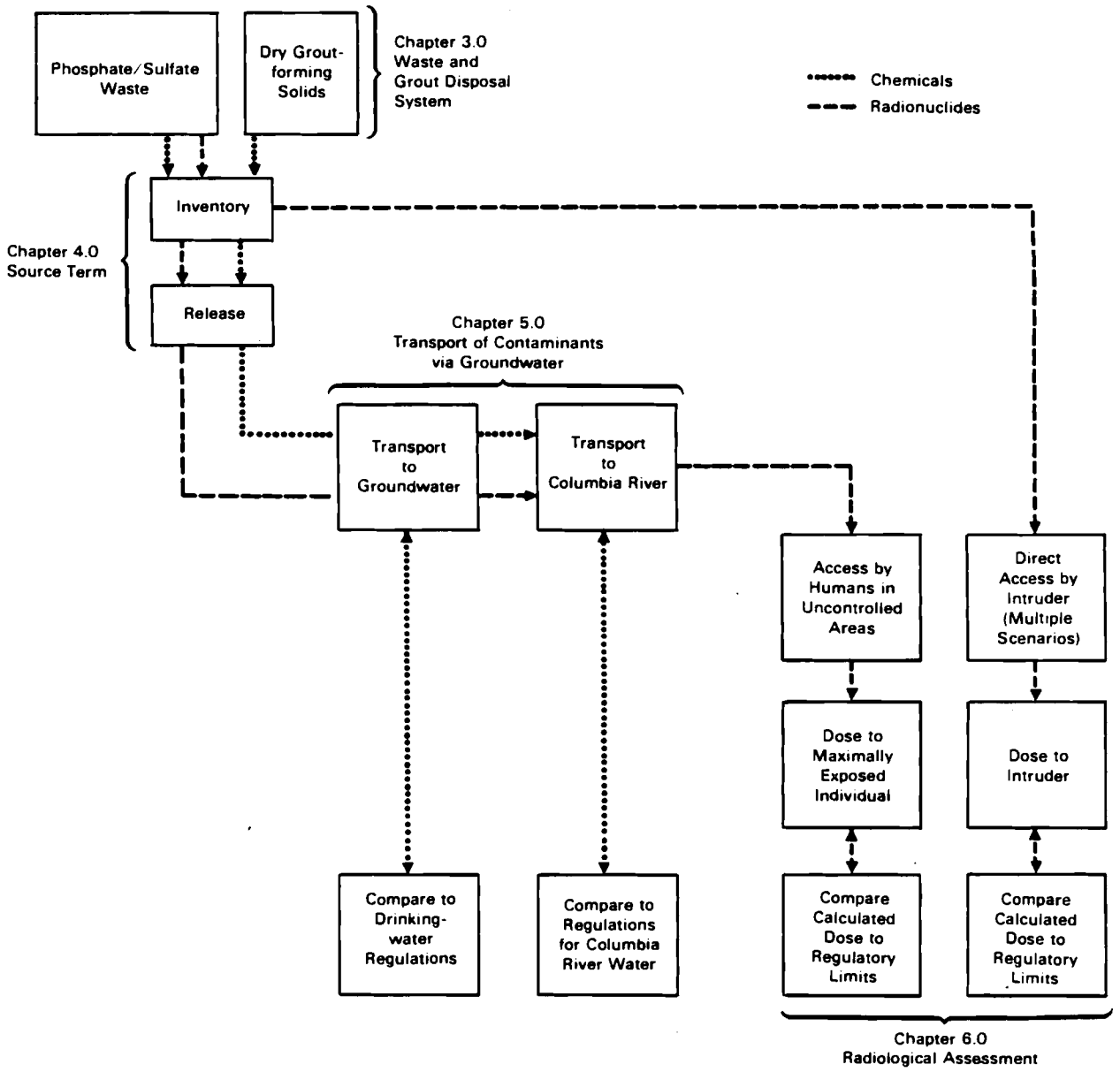


FIGURE 1.1. Progression and Organization of Long-Term Performance Assessment

2.0 PHILOSOPHY OF PERFORMANCE ASSESSMENT

This chapter begins by discussing the purpose of PAs, typical steps in the PA process, and how the results of a PA can be used. Section 2.2 gives an overview of how the PA process was used to evaluate a specific application: the performance of the disposal system for PSW grout.

2.1 GENERAL DISCUSSION OF PERFORMANCE ASSESSMENTS

Performance assessment is a multidisciplinary exercise aimed at predicting a system's response to reasonably projected conditions over a designated period of time. The primary purpose of a PA is to estimate whether a system, as designed, will comply with applicable regulations. Typically, long-term performance assessments are conducted for a range of performance conditions. Scenarios are selected that represent the expected behavior of the system as well as performance under stressed or partially failed conditions. Because researchers often lack quantitative data pertaining to the long-term durability of the system, conservative estimates are selected in an attempt to produce a bounding analysis (i.e., a prediction of performance under less than optimal conditions).

The steps of a PA are shown in Figure 2.1. First the system and its environmental setting are defined. Applicable regulations dictate the level of detail necessary to define the system. Second, a set of scenarios is developed to postulate reasonably foreseeable stresses on the system. Next, the physical responses of the system to the scenarios are modeled. Last, the consequences of the scenarios are calculated and compared to applicable regulations and design goals. If the consequences are acceptable, system planning may proceed.

Although the procedure for a PA is simple in concept, applying the approach to actual situations can be very difficult. This is particularly true for cases in which the engineered system is poorly defined, the environmental setting is not well known, or when the system's performance must be projected over extended time periods. Where unknowns or uncertainties exist, estimates must be made. As previously mentioned, conservative

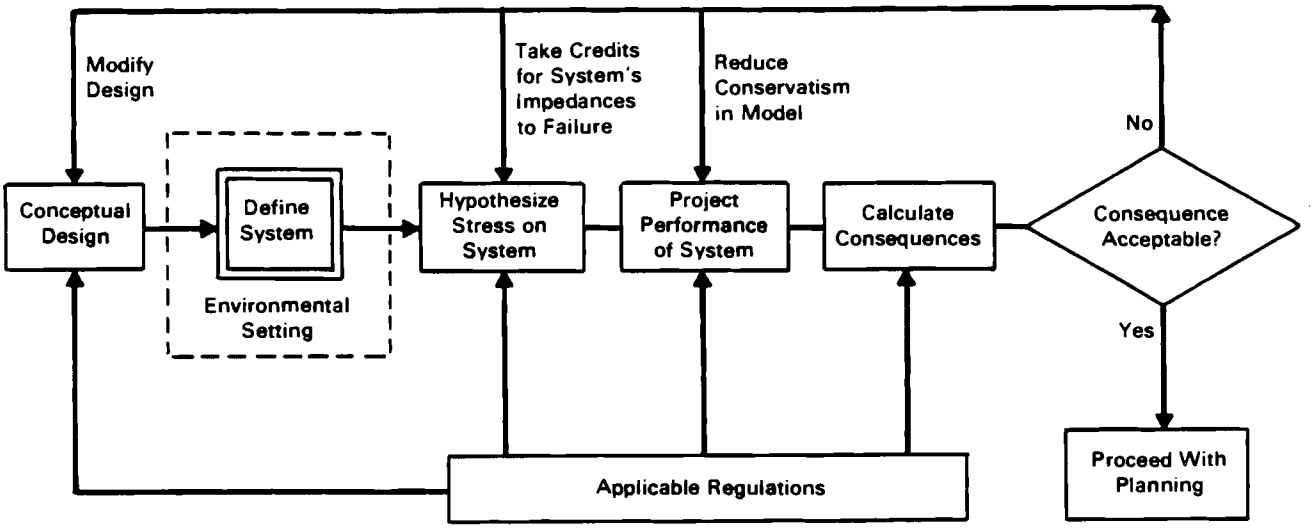


FIGURE 2.1. Procedure for a Performance Assessment

estimates are often used to ensure that the analysis will represent the upper range of credible impacts. For example, if we know that the rate of release for a certain contaminant is between x and y, with y representing the most rapid release, we model the release of the contaminant using the release rate y, so long as other model parameters are compatible with the release rate y.

Most PAs performed for waste disposal systems overestimate potential consequences in order to compensate for inherent uncertainties. Examples of these uncertainties are 1) waste characteristics, system design parameters, and waste form performance; 2) the understanding of contaminant transport; and 3) the selection of the reasonable intrusion scenarios to be analyzed.

2.2 APPROACH TO PERFORMANCE ASSESSMENT IN THIS REPORT

The purpose of this PA is to support the EA by investigating whether the PSW grout disposal system can adequately provide long-term protection of public health and safety. The PA contains a number of simplifying assumptions, as well as a level of uncertainty that cannot be quantified at this time. When the PSW grout disposal system and associated applicable regulations become more clearly defined, a "retroactive" PA could be conducted in order to more defensibly project regulatory compliance.

2.2.1 Overall Procedure

In this PA, we model the transport of contaminants from the waste form to points of human access. Following the general procedure that was shown in Figure 2.1, the PSW grout disposal system and its environmental setting were defined as shown in Figure 2.2. (A detailed description of the PSW grout disposal system is given in Section 3.3 of this report.) Because it is not yet possible to quantify the degree of protection afforded by the grout vault structure (i.e., cement walls and liners), the monolith was modeled as directly contacting the soil.

A number of scenarios were postulated to occur over a 10,000-year period following disposal of PSW grout. The scenarios identify pathways by which contaminants could be transported from the monoliths to humans. Two types of pathways were postulated: 1) migration of contaminants into the groundwater and then to the Columbia River, and 2) inadvertent intrusion into the grout site after a hypothetical loss of institutional (government) control. The first pathway (groundwater) applies both to radionuclides and nonradiological

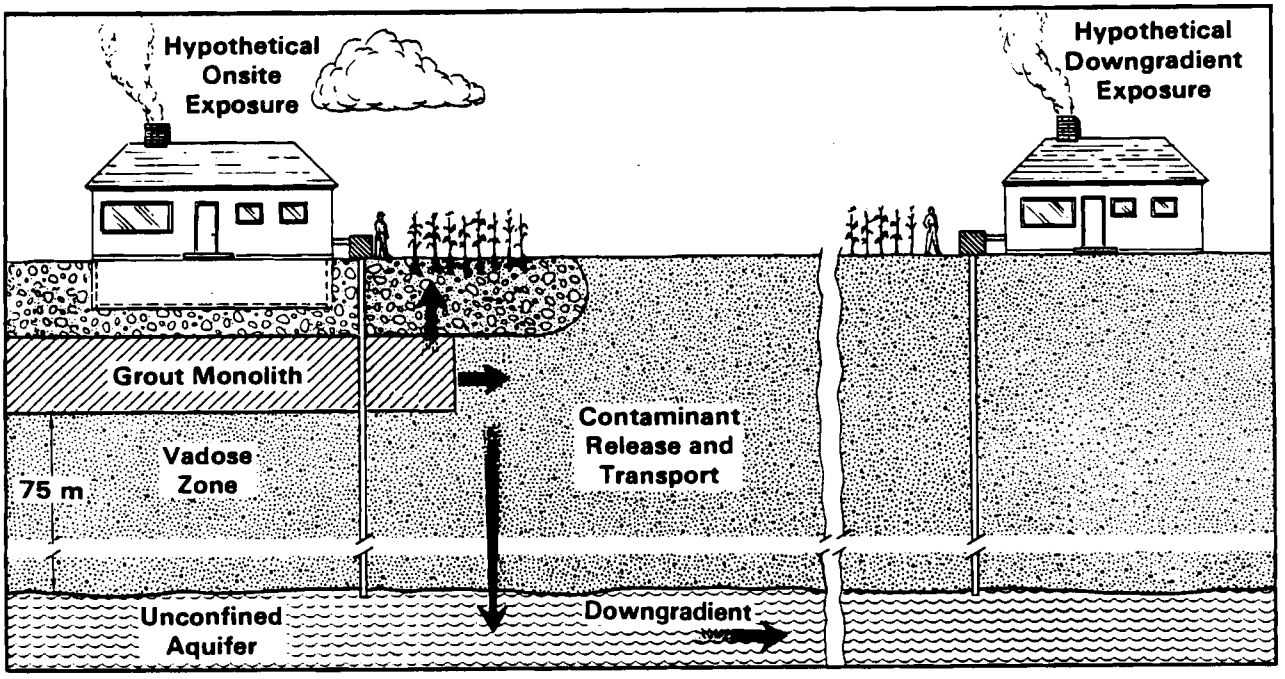


FIGURE 2.2. Conceptual Model for Grout Disposal System and Potential Exposure Pathways

chemicals. Contaminant transport via groundwater is described in Chapter 5.0. Calculated concentrations of chemical contaminants in groundwater and in the Columbia River were compared to corresponding regulations. However, concentrations of radionuclides must be translated to human radiation doses in order to compare them to regulations that limit radiation exposure to humans. Consequently, calculated concentrations of radionuclides in the groundwater and Columbia River were used as input for dose models, as described in Chapter 6.0. Chapter 6.0 also describes the second major pathway of contaminant transport: inadvertent intrusion into the grout disposal site. Calculated doses to an inadvertent intruder are presented.

2.2.2 Expected Performance Versus Modeled Performance

The model of the grout disposal system differs from the actual system in several ways. Figure 2.3 compares the system as modeled to the actual system. Detailed descriptions of the grout disposal system and modeling approach are contained in Chapters 3.0 through 6.0. However, a few general differences are noted here.

As designed, the grouted waste will be poured into plastic-lined, steel-reinforced concrete vaults. Each vault will be equipped with a leachate collection system. About 4 feet of nonradioactive grout will be poured on top of the radioactive grout. Above the grout will be a water-shedding cap and shielding material up to ground level. Over long periods of time, the liners and vault may fail, allowing contaminants to diffuse out of the vault and into the soil where they could be carried down into the aquifer by advecting water. Some contaminants would move more slowly because of geochemical interactions. The contaminants would become dispersed before and after reaching the aquifer, resulting in reduced concentrations. The monolith itself could also crack and crumble with age, potentially providing additional surface area from which contaminants could be leached.

As modeled, the grout monolith is bathed with incoming water. Because it is not yet possible to specify how or when its containment structures will fail, no containment credit was taken for the liners or vault. However, the monolith is assumed to remain intact (uncracked) over the long term. As

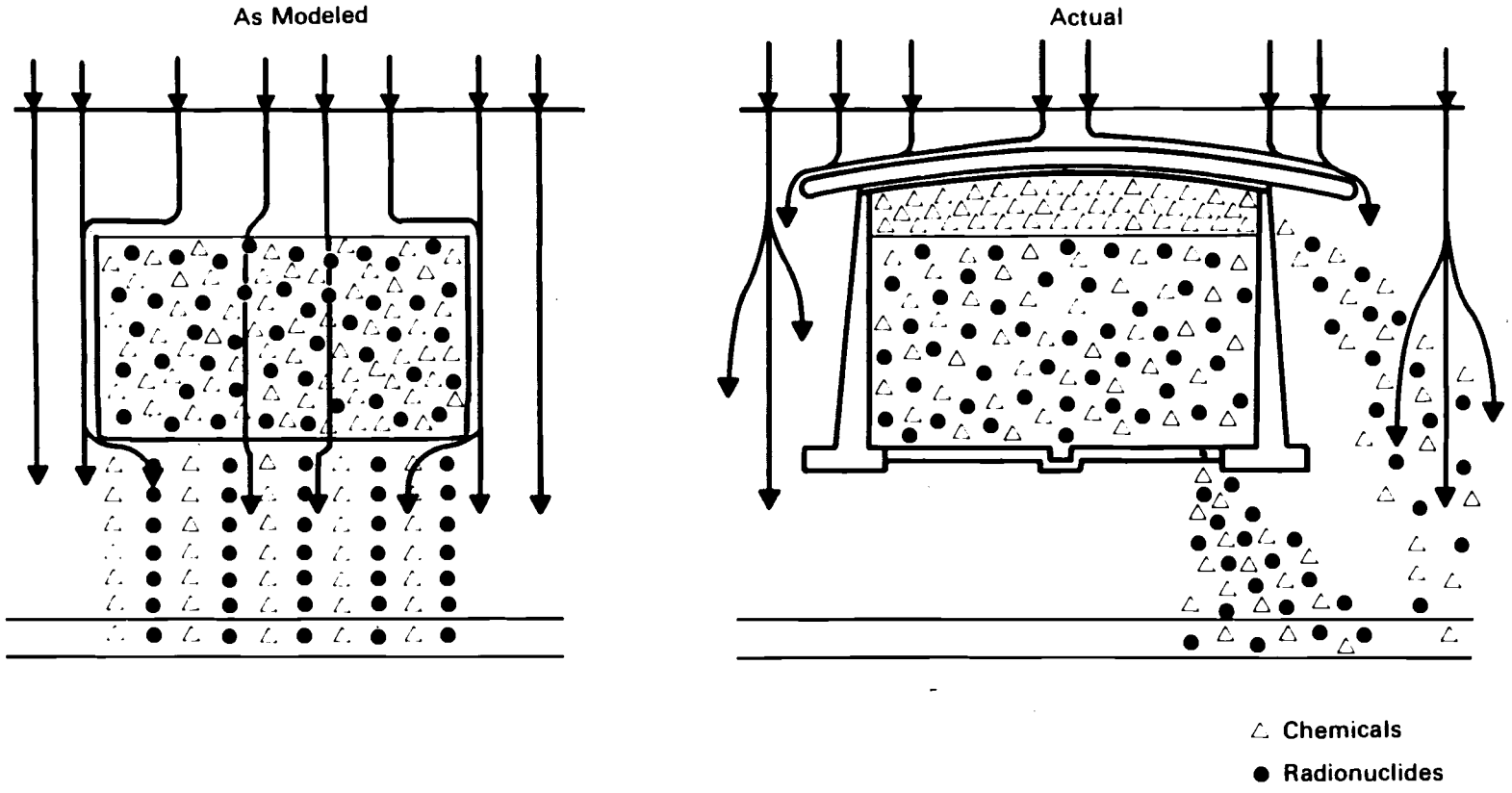


FIGURE 2.3. Comparison of Modeled Grout Disposal System to Actual System

modeled, the contaminants are uniformly distributed throughout the grout monolith, and the entire inventory is released at a constant rate into the soil below. No credit was taken for reduced concentrations due to dispersion. The contaminants are modeled as traveling directly down to the aquifer and then horizontally to a hypothetical well and on to the Columbia River.

3.0 DESCRIPTION OF WASTE AND GROUT DISPOSAL SYSTEM

A variety of low-level liquid wastes (LLW) can be immobilized by grouting. The specific low-level waste stream discussed in this report is called Phosphate/Sulfate Waste from N Reactor Operations (PSW). Section 3.1 discusses the nature and source of the waste streams that compose PSW. Section 3.2 identifies the estimated volumes of the waste before and after grouting, and the composition of the grouted waste. Section 3.3 describes the disposal system planned for the grouted waste.

3.1 ORIGIN OF PSW WASTE

Phosphate/Sulfate Waste is composed of three waste streams that originate at the N Reactor in 100-N Area of the Hanford Site. (The N Reactor produces special nuclear materials, and its byproduct steam is used to generate electricity.) The waste streams are designated as phosphate waste, sulfate waste, and sandfilter backwash.

Phosphate waste is produced during periodic reactor decontamination using a commercial decontamination agent that contains phosphoric acid, citric acid, and trace amounts of other chemicals. The resulting waste is a dilute aqueous solution of trisodium phosphate and citrate, containing trace amounts of inorganic chemicals and radioactive elements (radionuclides) of varying half-lives.

The two other waste streams that make up PSW--sulfate waste and sandfilter backwash--are produced during periodic cleanup of the water in N Reactor's spent-fuel storage basin. Ion-exchange resins remove radionuclides from recirculated storage basin water. Sulfuric acid is used to regenerate the cation exchange resin; sodium hydroxide is used to regenerate the anion exchange resin. Extensive rinsing of the resins produces a dilute sodium sulfate waste solution. The acidic solution is adjusted to a pH of 12 with sodium hydroxide to limit corrosion of the carbon-steel storage tanks. Also present in the basin-water cleanup system is a sandfilter for removing entrained solids from the water. Periodic flushing of the filter with water generates sandfilter backwash waste in the form of a dilute slurry.

3.2 VOLUME AND COMPOSITION OF PSW STREAM

The volume of PSW to be grouted is projected to be about 43,000 cubic meters. These 43,000 cubic meters are expected to be composed of 2.45×10^4 cubic meters of sulfate waste and 1.14×10^4 cubic meters of phosphate waste, which are then diluted with 7.2×10^3 cubic meters of flush water. (The sandfilter backwash sludge is included in the volume of sulfate waste.) For each gallon of PSW, approximately 7.5 pounds of grout-forming solids (cement, fly ash, and clay) will be added. The volume of the grouted waste will be approximately 56,000 cubic meters; the density will be about 12 lb/gal (1.44×10^3 kg/m^3). The proportions of each component in the dry solids blend are shown in Table 3.1.

Concentrations of radionuclides and chemicals present in PSW are listed in Table 3.2. Not all radionuclides are significant in the assessment of the long-term performance of disposed grout. Certain radionuclides have sufficiently short half-lives such that they do not affect the results of the PA. Other species can be considered "key" radionuclides because of their concentrations, longevity, and mobility. The chemicals listed here are those inorganic species regulated by the Washington State Department of Social and Health Services (WAC 1985).

Nitrogen is regulated when it exists in the form of nitrate (NO_3). However, Table 3.2 includes nitrogen from ammonia (NH_3). Nitrite and organic

TABLE 3.1. Typical Dry Solids Blend[(a)]

Material	Amount (wt%)
Type I-II-LA portland cement	41
Centralia, WA ASTM Class F, fly ash	40
Attapulgite-150 drilling clay	11
Indian Red pottery clay	8

(a) Based on the formulation developed at Oak Ridge National Laboratory.

TABLE 3.2. Concentrations of Chemicals and Radionuclides Present in PSW Streams[a]

Chemical	Concentration in Sulfate Stream, moles/L	Concentration in Phosphate Stream, moles/L	Concentration in Sandfilter Backwash, µg/g
Arsenic (As)	6.7×10^{-8}	6.7×10^{-8}	0.08
Barium (Ba)	$<4.0 \times 10^{-7}$	6.3×10^{-7}	4
Cadmium (Cd)	$<2.2 \times 10^{-6}$	3.6×10^{-8}	1.6
Chromium (Cr)	$<6.0 \times 10^{-7}$	4.7×10^{-5}	1.3
Fluoride (F)	$<1.4 \times 10^{-4}$	1.0×10^{-3}	2.5
Lead (Pb)	$<3.9 \times 10^{-6}$	3.6×10^{-7}	32
Mercury (Hg)	1.0×10^{-7}	4.8×10^{-8}	0.5
Selenium (Se)	6.3×10^{-8}	6.3×10^{-8}	0.5
Silver (Ag)	$<9.0 \times 10^{-7}$	1.9×10^{-7}	32
Chloride (Cl)	1.1×10^{-3}	1.0×10^{-3}	2.5
Copper (Cu)	$<5.0 \times 10^{-7}$	1.8×10^{-6}	5.9
Iron (Fe)	$<1.6 \times 10^{-6}$	5.6×10^{-3}	1,320
Manganese (Mn)	$<3.6 \times 10^{-5}$	6.5×10^{-4}	8.2
Sulfate (SO_4)	2.4×10^{-2}	1.3×10^{-3}	25
Zinc (Zn)	$<8.0 \times 10^{-7}$	1.8×10^{-5}	31
Nitrate (NO_3)	2.2×10^{-4}	$<2.0 \times 10^{-3}$	---
Ammonia (NH_3)	---	$<8.7 \times 10^{-2}$	---

Radionuclide[a]	Concentration in Sulfate Stream, Ci/L	Concentration in Phosphate Stream, Ci/L	Concentration in Sandfilter Backwash, Ci/kg
Carbon-14	9.9×10^{-14}	6.0×10^{-8}	8.4×10^{-15}
Cobalt-60	2.0×10^{-6}	1.6×10^{-4}	6.2×10^{-1}
Strontium-90	3.3×10^{-5}	2.2×10^{-10}	2.8×10^{-6}
Technetium-99	4.0×10^{-9}	4.0×10^{-9}	4.2×10^{-8}
Iodine-129	7.0×10^{-14}	1.1×10^{-15}	5.9×10^{-15}
Cesium-137	5.0×10^{-5}	9.3×10^{-7}	3.8×10^{-2}
Uranium-238	1.1×10^{-9}	1.8×10^{-11}	1.1×10^{-12}
Plutonium-239	2.4×10^{-7}	5.2×10^{-9}	1.0×10^{-7}
Amercium-241	3.3×10^{-10}	1.2×10^{-8}	6.2×10^{-7}

(a) Source: U.S. DOE (1986a).

nitrogen are not present in the waste streams. Including all sources of nitrogen in the inventory is consistent with the Environmental Protection Agency's (EPA's) Proposed Rule for National Primary Drinking Water Regulations, which states, "Most nitrogeneous materials in natural waters tend to be

converted to nitrate, and, therefore, all sources of combined nitrogen (particularly organic nitrogen and ammonia) should be considered as potential nitrate sources" (U.S. EPA 1985b).

The state drinking water standards do not apply directly to the groundwater beneath the Hanford Site because it is not a continuous source of public drinking water. However, the chemical species evaluated in this PA were selected from those regulations for comparison purposes. Comparison with these values provides an indication of the grout system's performance under the most restrictive of standards.

3.3 DISPOSAL STRUCTURE

The slurry will be pumped into subsurface disposal vaults, where it will harden into a solid matrix that immobilizes the waste. A preliminary design of the disposal vault is shown in Figure 3.1. Each vault will be 10.4 m deep, 15.24 m wide, and 38.1 m long. The walls and floor will be made from steel-reinforced concrete. Each vault will be lined with a high-density polyethylene liner which will prevent dewatering of the grout during setting and curing. An additional liner will provide redundant protection. Two leachate collection systems will be installed: one to remove leachate from the base of the grout monolith; the other from between the two liners at the bottom of each vault (to remove any leachate that may penetrate the primary liner). A concrete cover and a clay or asphalt cap will then be placed over the vault, and the remaining volume above the cap will be filled in with shielding backfill up to grade.

Each vault will hold approximately 5,000 cubic meters of radioactive grouted waste. The unfilled volume of the vault (about 1,000 cubic meters) will be filled with nonradioactive grout to prevent subsidence. Hence, the radioactive grout will be about 9 m thick, and the nonradioactive grout will be about 1.5 m thick. The final distance from ground level down to the top of the radioactive grout will be 5 m.

Final closure of the PSW grout disposal site will include emplacement of an interim surface barrier over the vaults.

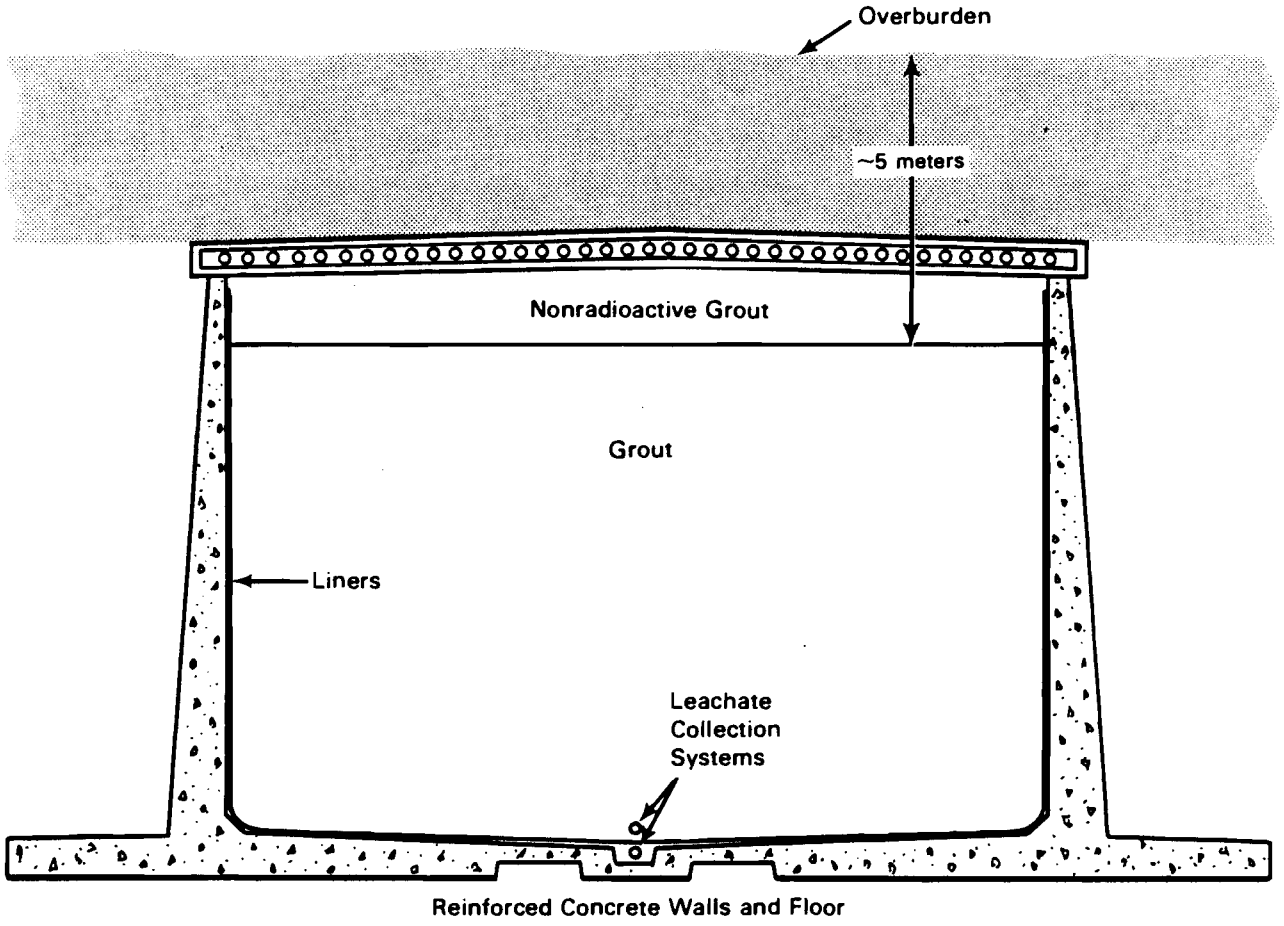


FIGURE 3.1. Preliminary Design of a PSW Grout Disposal Vault

4.0 SOURCE TERM

To quantitatively project a contaminant's movement out of a grout monolith and through the environment, the source term of each contaminant in the grout monolith must be defined. In this context, "source term" refers not only to the inventory of contaminants in the grout monolith, but also to the amount released from the monolith and hence available for transport. This chapter describes how the source terms for specific contaminants were calculated for the PA for PSW grout.

Section 4.1 discusses the inventories for the radionuclides and the regulated nonradioactive chemicals. Section 4.2 describes the calculation of the fraction of waste released from the monolith as a function of time.

4.1 CALCULATING THE WASTE INVENTORIES

The inventories of specific chemicals and radionuclides in the PSW streams are based on analyses of actual waste samples. The dry grout-forming solids added to the waste also contribute to the total inventory of regulated nonradioactive contaminants. Estimates of the dry solids' contribution to this inventory are based on quantitative analyses of various dry solids components. Sections 4.1.1 and 4.1.2 describe how the inventory of each chemical and radionuclide was calculated.

4.1.1 Chemical Inventory

As stated in Section 3.2, the nonradioactive chemicals in PSW grout that were evaluated in the PA were those listed in state drinking-water standards (WAC 1985). The primary and secondary standards are included. The inventory of all regulated inorganic chemicals in PSW grout (Table 4.1) includes contributions from the three waste streams, the dry grout-forming solids, and any processing additives such as de-aerating agents. The contributions from the waste streams are projected to be the same as those measured in actual waste samples. The chemical concentrations in PSW streams shown in Table 3.2 were multiplied by the total volume of each waste stream to produce the data shown

TABLE 4.1. Nonradiological Regulated Chemicals in PSW Grout, kg

Chemical	Source					
	From Phosphate Stream[a]	From Sulfate Stream[a]	From Sandfilter Backwash[a]	Subtotal	From Dry Solids[b]	Total[b]
Arsenic	0.057	0.123	0.0001	0.180	680	680
Barium	0.980	1.35	0.005	2.3	54,000	54,000
Cadmium	0.046	6.06	0.002	6.1	64	70
Chromium	88	0.764	0.0016	29	4,300	4,300
Fluoride	223	65.1	0.003	290	---	290
Lead	0.843	19.8	0.038	21	45	66
Mercury	0.109	0.49	0.0006	0.61	4.5	5.1
Selenium	0.057	0.122	0.0006	0.18	45	46
Silver	0.230	2.38	0.038	2.6	45	48
Chloride	412	955	0.003	1,400	14,000	15,000
Copper	1.3	0.780	0.007	2.1	2,000	2,000
Iron	3,550	2.19	1.58	3,600	1,900,000	1,900,000
Manganese	408	48.5	0.010	460	21,000	21,000
Sulfate	1,400	56,500	0.030	58,000	1,100,000	1,200,000
Zinc	13.5	1.28	0.037	15	5300	5,300
Nitrogen	14,000[c]	75	0	14,000	---	14,000

(a) Source: U.S. DOE 1986(a).
(b) Includes chemicals in the 1.5-m layer of nonradioactive grout.
(c) Essentially all in the form of ammonia.

in Table 4.1. These volumes were 25,500 cubic meters of sulfate waste, 11,400 cubic meters of phosphate waste, and 1,200 kilograms of sandfilter backwash.

Chemicals in the Dry Solids

The remainder of this section describes how the chemical contributions from the dry grout-forming solids were calculated. Contributions from the dry solids were estimated from three sources: 1) analyses of the four dry solids, 2) a separate analysis of the Centralia fly ash, and 3) measurements of chemicals in an actual sample of grout. In the last set of measurements, the waste stream used to make the grout sample was 60% actual phosphate waste and 40% simulated sulfate waste (Serne et al. 1986). The processing additive for PSW, tributyl phosphate, is not a regulated chemical and hence does not contribute to the calculated inventory of chemicals.

The chemical contributions from the dry solids were calculated as follows. When more than one measurement of a chemical was available, the higher value was used. Certain chemicals were not detected in the samples, and hence their detection limits were used in calculating the total inventory. Using detection limits is conservative because it can result in the calculated inventory being higher than the actual inventory.

The total amount of dry solids in PSW grout is calculated in Equation (1):

$$\left(43{,}000\ m^3\ \text{waste}\right)\left(\frac{7.5\ \text{lb solids}}{\text{gal waste}}\right)\left(\frac{1\ \text{gal}}{0.00378\ m^3}\right)\left(\frac{0.454\ \text{kg}}{\text{lb}}\right) = \begin{matrix}3.88 \times 10^7\ \text{kg}\\ \text{solids in}\\ \text{PSW grout}\end{matrix} \quad (1)$$

The dry solids in the 1.5 m of nonradioactive grout to be placed over the 9-m-thick PSW grout must also be accounted for. Assuming the nonradioactive grout will be made from the same solids that will be used to produce the waste grout, we added it to the PSW grout volume: $(3.88 \times 10^7\ \text{kg})(10.5\ \text{m}/9\ \text{m}) = 4.5 \times 10^7\ \text{kg}$ dry solids.

An analysis of the four dry solids included measurements of arsenic, barium, chromium, iron, manganese, sulfur, and zinc (Table 4.2). These values were multiplied by their weight fractions in the dry blend for grout (see

TABLE 4.2. Concentrations of Chemicals in Dry Grout-Forming Solids

Element	Cement	Fly Ash	Attapulgite Clay	Pottery Clay
Arsenic (ppm)	6.3 ± 2.5	23 ± 2	<2	15.5 ± 1.5
Barium (ppm)	970 ± 85	1540 ± 90	330 ± 100	600 ± 100
Chromium (ppm)	<80	75 ± 20	120 ± 20	90 ± 20
Iron (wt%)	3.48 ± 0.17	4.73 ± 0.23	2.09 ± 0.10	4.38 ± 0.20
Manganese (ppm)	365 ± 15	590 ± 20	375 ± 10	203 ± 8
Sulfur (wt%)	1.1 ± 0.4	0.28 ± 0.1	<0.03	<0.03
Zinc (ppm)	50 ± 6	180 ± 10	82 ± 5	85 ± 5

Table 3.1), and then multiplied by the total kilograms of dry solids. The calculation of arsenic is reproduced here as an example:

$$[(8.8 \text{ ppm arsenic in cement})(0.41) + (25 \text{ ppm in flyash})(0.40) + (2 \text{ ppm in attapulgite})(0.11) + (17 \text{ ppm in pottery clay})(0.08)] \times 10^{-6} \times 4.5 \times 10^{7} \text{ kg solids} \quad (2)$$

$$= 680 \text{ kg arsenic in dry solids}$$

Inventory of Sulfate. In the PA, the total amount of elemental sulfur measured in the dry solids was conservatively assumed to be in the form of sulfate (SO_4), which is a regulated chemical. Using the same procedure, approximately 1×10^6 kg of sulfate is calculated to exist in all PSW grout; the dry solids contribute about 95%. In a separate analysis, 11,000 to 22,000 ppm of sulfate were measured in a grout sample (Serne et al. 1986). The higher measurement in that analysis yields the same mass of sulfate as was predicted from the measurement of sulfur in dry solids:

$$\left(\frac{22{,}000 \text{ kg sulfate}}{10^6 \text{ kg dry grout}}\right)\left(\frac{1 \text{ kg dry grout}}{2 \text{ kg wet grout}}\right)\left(56{,}000 \text{ m}^3 \text{ grout}\right)\left(\frac{1.44 \times 10^3 \text{ kg grout}}{\text{m}^3 \text{ grout}}\right)\left(\frac{10.5 \text{ m nonradioactive grout + waste grout}}{9 \text{ m waste grout}}\right) = 1 \times 10^6 \text{ kg sulfate in PSW grout} \quad (3)$$

Therefore, because two independent calculation methods yielded the same value, we have greater confidence that the calculated value is accurate.

Inventory of Chloride. Chloride is assumed to exist in the dry solids in a concentration of <300 ppm, based on a measurement of chloride in the grout sample (Serne et al. 1986). This assumption is conservative because it does not discount the amount of chloride contributed by the PSW. The calculation of chloride is parallel to the calculation of sulfate in Equation (3).

Database. A sample of Centralia fly ash was also analyzed by California Analytical Laboratories, Inc. The results of this analysis are shown in Table 4.3. These data were used to enhance the database of chemical concentrations in fly ash. When the database contained more than one measurement of a certain chemical, the higher value was used in subsequent calculations.

No data were available for the concentrations of cadmium, lead, mercury, selenium, silver, or copper in the other dry solids (cement, attapulgite clay,

TABLE 4.3. Concentrations of Chemicals in Centralia Fly Ash[a]

Element	Concentration, ppm
Arsenic	<2
Barium	290
Cadmium	1.4
Chromium (Total)	25
Lead	<1
Mercury	<0.1
Selenium	1
Silver	<1
Copper	43
Zinc	41

(a) Letter, D. J. Leu (Department of Health Services, State of California) to T. A. Fox (Pozzolanic International, Mercer Island, Washington), August 20, 1984.

and Indian Red pottery clay). However, the concentrations of these trace elements are not expected to exceed their concentrations in fly ash. Therefore, the concentrations of these elements as measured in fly ash were assumed to be present in all 4.5×10^7 kilograms of dry solids to be used in PSW grout.

No data were available on the concentrations of fluoride or nitrogen in the four dry solids. However, a higher concentration of fluoride in PSW grout would not change the projected concentrations of fluoride in groundwater because fluoride is assumed to reach a maximum concentration in the vadose zone[(a)] due to solubility constraints (see Section 4.2.2). Therefore, any additional fluoride would form precipitates rather than dissolving in the groundwater. The calculated inventory of nitrogen is thought to be conservative because it includes nitrogen actually present as ammonia in addition to nitrogen in its regulated form, nitrate.

4.1.2 Radionuclide Inventory

The radionuclides addressed in this long-term PA are only those that have significant half lives and significant environmental impacts. Using the concentrations listed in Table 3.2, the total curies in PSW grout were calculated as shown in Table 4.4. The calculations are based on 24,500 cubic meters of

TABLE 4.4. Radionuclides in PSW Grout[(a)]

Radionuclide	Sulfate, Ci	Phosphate, Ci	Sandfilter Backwash, Ci	Total Ci	Concentration in Grout, Ci/m^3
Carbon-14	2.4×10^{-6}	6.8×10^{-1}	1.0×10^{-11}	6.8×10^{-1}	1.2×10^{-5}
Cobalt-60	4.9×10^{1}	1.9×10^{3}	7.4×10^{2}	2.7×10^{3}	4.8×10^{-2}
Strontium-90	8.1×10^{2}	2.5×10^{-3}	3.4×10^{-3}	8.1×10^{2}	1.4×10^{-2}
Technetium-99	9.8×10^{-2}	4.5×10^{-2}	5.1×10^{-5}	1.4×10^{-1}	2.6×10^{-6}
Iodine-129	1.7×10^{-6}	1.2×10^{-8}	7.1×10^{-12}	1.7×10^{-6}	3.1×10^{-11}
Cesium-137	1.2×10^{3}	1.1×10^{1}	4.6×10^{1}	1.3×10^{3}	2.3×10^{-2}
Uranium-238	2.7×10^{-2}	2.0×10^{-4}	1.3×10^{-9}	2.7×10^{-2}	4.8×10^{-7}
Plutonium-239	5.9×10^{0}	5.9×10^{-2}	1.2×10^{-4}	5.9×10^{0}	1.1×10^{-4}
Americium-241	8.1×10^{-3}	1.4×10^{-1}	7.4×10^{-4}	1.5×10^{-1}	2.6×10^{-6}

(a) 5.66×10^4 m^3 of disposed grout. Source: U.S. DOE 1986(a).

(a) The vadose zone is the unsaturated region of soil between the ground surface and the water table.

sulfate waste, 11,400 cubic meters of phosphate waste, and 1,200 kilograms of sandfilter backwash. The final concentrations are also based on the current projection of 5.66×10^4 cubic meters of grout at final disposal.

Radionuclides could also be present in fly ash, one of the dry grout-forming solids. Measurable levels of potassium-40 and radium-226 are known to be present in certain certain fly ashes (up to 26 pCi/g of ^{40}K and up to 10 pCi/g of ^{226}Ra), as reported by the Electric Power Research Institute (EPRI 1983). However, we did not attempt to quantify the radiological contribution, if any, from Centralia fly ash in this analysis. A radiochemical assay of Centralia fly ash is planned, and if the activity is found to differ significantly from that naturally present in soil, the results could be used to modify this analysis.

4.2 RELEASE OF WASTE FROM GROUT MONOLITH

Release from the grout monoliths was modeled in two ways dependent upon the chemical speciation of the contaminant under consideration. Contaminants that are not limited by solubility constraints were modeled as being released at a constant rate while the less soluble contaminants were assumed to be released according to their solubility limits and the annual volume of water available for transport. The subsequent transport of the contaminants in the vadose zone and in the groundwater is described in Chapter 5.0.

4.2.1 Constant-Release Model

Contaminant movement downward from the PSW grout disposal site was modeled as constant across a horizontal cross-section beneath the disposal site. This implies a uniform release per horizontal area of soil in the grout site. We assume that the grout monoliths are uniformly spaced throughout the grout site, and that the composition of each monolith is the same. For the soluble contaminants, the release rate for each monolith is modeled as a single constant value, unaffected by the water flow rate or water chemistry. The constant release continues until the total radionuclide and chemical inventory in the monolith has been leached.

Calculating the Period of Total Release

The model calculations use 10,000 years as the release time interval for release of all radionuclides and chemicals from a PSW grout monolith (with the exception of those chemicals with solubility-controlled releases). This release interval was calculated using laboratory data from PNL leaching experiments conducted on small PSW grout cylinders. The following equation, which is supported by the American Nuclear Society (ANS 1984), describes a fractional release from cylindrical monoliths that is controlled by diffusion of contaminants through the grout (Huizenga et al. 1986):

$$FR = 2\ (A/V)\ (D_e\ t/\pi)^{1/2} \qquad (4)$$

where FR = fraction released
A = geometric surface area of the cylinder (cm^2)
V = geometric volume of the cylinder (cm^3)
D_e = effective diffusion coefficient of the contaminant in grout (cm^2/sec)
t = time (sec)

It is assumed that the contaminants are initially homogeneously dispersed in the cylinder, so that the release rate of any contaminant from any part of the surface of the cylinder is the same. It is also assumed that the release from a cylinder of finite size can be approximate by the release from a semi-infinite slab (which never completely depletes), and that the leached contaminants are swept away rapidly such that the solution concentration at the grout surface is always zero for each contaminant. These assumptions maximize the concentration gradient between the monolith and the surrounding soil, and, as a result, maximize the projected rate of diffusional release.

Data from the PNL laboratory studies (Serne et al. 1986) were plotted as fraction released versus time, and a value for the effective diffusion coefficient (D_e) was estimated from a best fit of Equation (4) to the data. A D_e value of approximately 10^{-9} cm^2/sec was obtained for the most mobile species. It was then assumed that the model could be applied to release of contaminants from buried grout monoliths by using the same D_e and scaling the time axis based on differences in the surface-area-to-volume ratio of the two grout

matrices. The surface-area-to-volume ratio for the large monolith (10.4 m x 15.24 m x 38.1 m) is 3.8×10^{-3} cm^{-1}.

Although the geometry of the lab tests differs from that of the actual monoliths, (elongated rectangles rather than cylinders), scaling diffusional release based on the surface-area-to-volume ratio has been shown to be a close approximation when less than 20% of the total inventory has been released. As discussed below, the release rates in this PA were based on the fraction released during the first 70 years after disposal. This fraction is 0.7%, which is significantly below the 20% upper bound. Contaminant release that is controlled strictly by diffusion is not constant with time; rather, it is faster at the beginning of release when the concentration gradient is the highest. For this PA, the release rate was based on an estimation of the amount of a contaminant that would be released from a grout monolith during a person's average lifetime (70 years). Because the concentration gradient is highest initially, the fastest release would occur during the first 70 years after disposal.

The rate of contaminant release was estimated by using Equation (4) with a D_e of 10^{-9} cm^2/sec and a surface-area-to-volume ratio for the large grout monolith of 3.8×10^{-3} cm^{-1}. As mentioned above, the total amount of contaminant that would be released by diffusion in the first 70 years is approximately 0.7% of the inventory. This 70-year average release rate was used to extrapolate a constant release curve from zero to the 100% released limit (FR = 1) as shown in Figure 4.1. This limit was reached at approximately 10,000 years. (The initial parts of the curves are not drawn to scale, in order to show the detail at the 70-year point.) Throughout this analysis, an average release rate of approximately 0.01% per year was assumed, based on the initial 70-year time period of diffusioned release.

Release Rate and Monolith Cracking

The assumed constant release rate does not take into account a change in the rate of release caused by cracking of the grout monolith. It is difficult to project how cracking may affect contaminant migration. Models can be postulated to project how cracking may be detrimental or beneficial. For example, impedances to contaminant diffusion may develop as a result of dry

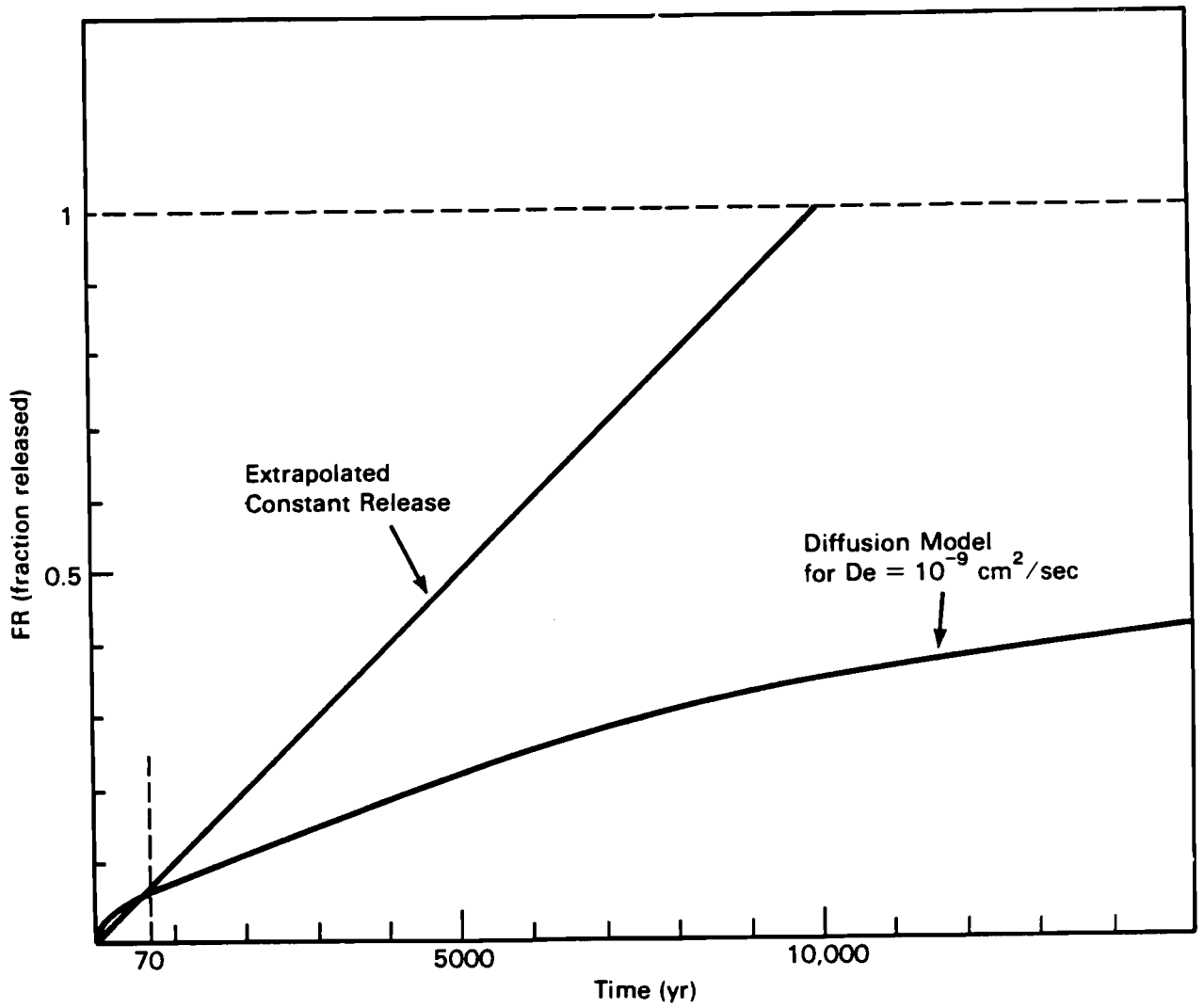


FIGURE 4.1. Scaled Curve of Fraction of Contaminant Released from Grout Monolith as a Function of Time, Showing Extrapolation Method for Constant Release Approximation

inner cracks within the monolith. On the other hand, if water were to penetrate the grout vault and flow between the cracks, migration of contaminants would be enhanced because leach rates are directly related to the surface-area-to-volume ratio. Preliminary leach tests performed on crushed grout samples indicate a disproportionately low rate of release based on scaling up Equation (4) by the surface-area-to-volume ratio (Serne et al. 1986). At this point, it is not possible to ascertain the reason for this anomaly. The boundary conditions of the crushed grout experiments are not consistent with

those used when developing Equation (4); therefore, it is not strictly a valid expression for describing release from the crushed grout by passive diffusion. Another possible explanation for the lower-than-expected release of contaminants from crushed grout is that release is not strictly caused by passive diffusion in the grout pore water. It is possible that chemical reactions occur on the internal surfaces of the grout or in the pore water, thus altering contaminant transport. Such chemical reactions have been evidenced by the presence of a heavy calcium carbonate precipitation on the crushed grout samples. Studies are continuing in an attempt to validate models that predict release in cases where the waste form is stressed and cracked. As this information becomes available, it can be used to update this and future analyses.

4.2.2 Solubility-Controlled Release

Solubility constraints are expected to limit the rates of release of certain chemical contaminants from the grout matrix to the water in the vadose zone. To determine which of the chemicals in PSW grout are solubility-controlled, site-specific parameters of the soil pore water outside of the grout vaults were used as input to the geochemical computer code MINTEQ (Felmy, Girvin and Jenne 1984). Based on the results, ten of the PSW chemical species regulated in WAC-248-54 (WAC 1985) were assumed to have solubility-controlled release rates: barium, cadmium, chromium, copper, iron, lead, manganese, silver, fluoride, and zinc. The solubility limits (maximum predicted concentrations in the vadose-zone water) and underlying parameters are shown in Table 4.5.

Concentrations of these elements in the groundwater system were estimated to be at their maximum solubilities in the vadose-zone water. Then, as contaminants in the soil water percolated downward and mixed with water flowing in the unconfined aquifer beneath the PSW grout disposal site, concentrations of contaminants were reduced by the dilution.

TABLE 4.5. Solubility-Limited Concentrations as Calculated by MINTEQ

Element	Controlling Solid	Maximum Predicted Concentration, mg/L
Manganese	$MnHPO_4$	0.02
Iron	$Fe_3(OH)_8$	0.025
Chromium	$Cr(OH)_3$	0.09
Copper	CuO (tenorite)	0.06
Barium	$BaSO_4$ (barite)	0.04
Zinc	Zn_2SiO_4 (willemite)	0.02
Cadmium	$CdCO_3$ (otavite)	0.002
Silver	AgCl (cerargyrite)	0.075
Lead	$PbCO_3$ (cerussite)	0.29
Fluoride	CaF_2 (fluorite)	2.04

Assumed Parameters

pH, 8.1
Eh, 295 mV
Potassium, 7.8 mg/L
Sodium, 25 mg/L
Magnesium, 14.4 mg/L
Calcium, 56 mg/L
Chloride, 22 mg/L
Sulfate, 85 mg/L
H_4SiO_4, 54 mg/L
total carbonate, 86 mg/L

5.0 TRANSPORT OF CONTAMINANTS VIA GROUNDWATER

This chapter presents the conceptual model of the groundwater scenario, and describes the mathematical theory on which it is based. Numerical values of input parameters required by the model are presented, as well as assumptions and approximations that were made when actual physical data was unavailable. Section 5.1 gives an overview of the model. Sections 5.2 and 5.3 explain the two submodels used to simulate the groundwater scenario: water flow and contaminant transport in the vadose zone and in the unconfined aquifer, respectively.

5.1 GROUNDWATER SCENARIO--OVERALL CONCEPTUAL MODEL

Figure 5.1 illustrates the overall conceptual model for the groundwater scenario. In this scenario, impacts to members of the general public are postulated to occur through the following pathway: contaminants (mobile radionuclides and nonradioactive regulated chemicals) migrate out of the grout vaults over a period of time. No credit is taken for vault containment. The contaminants are assumed to enter the soil solution (moisture present in the soil) by the mechanisms described in Section 4.2. The contaminants then move downward through the vadose zone to the unconfined aquifer beneath the grout site.[(a)] Some radionuclides sorb to the porous materials in the ground and decay into insignificant quantities as they move downward. Solubility-limited chemicals are postulated to exist in their maximum concentrations in the vadose zone. Upon entering the unconfined aquifer, the radionuclides and chemical contaminants move into the groundwater that is flowing under the grout site. The contaminants are modeled as migrating along with the groundwater as it moves past the location of a hypothetical well and then finally on to the Columbia River.

(a) Contaminants in this scenario are assumed to have vapor pressures so low that vapor phase transport is negligible. The scenario further assumes that there are no mechanisms by which contaminants may migrate upward and be lost to the atmosphere through volatilization, or be removed from the site by soil erosion or water runoff. Exposure via a direct air pathway is addressed, however, by the inadvertent intrusion scenarios described in Chapter 6.0. The impacts resulting from inhalation of resuspended contaminants that were mixed directly with the soil are expected to be bounded by the scenarios in Chapter 6.0.

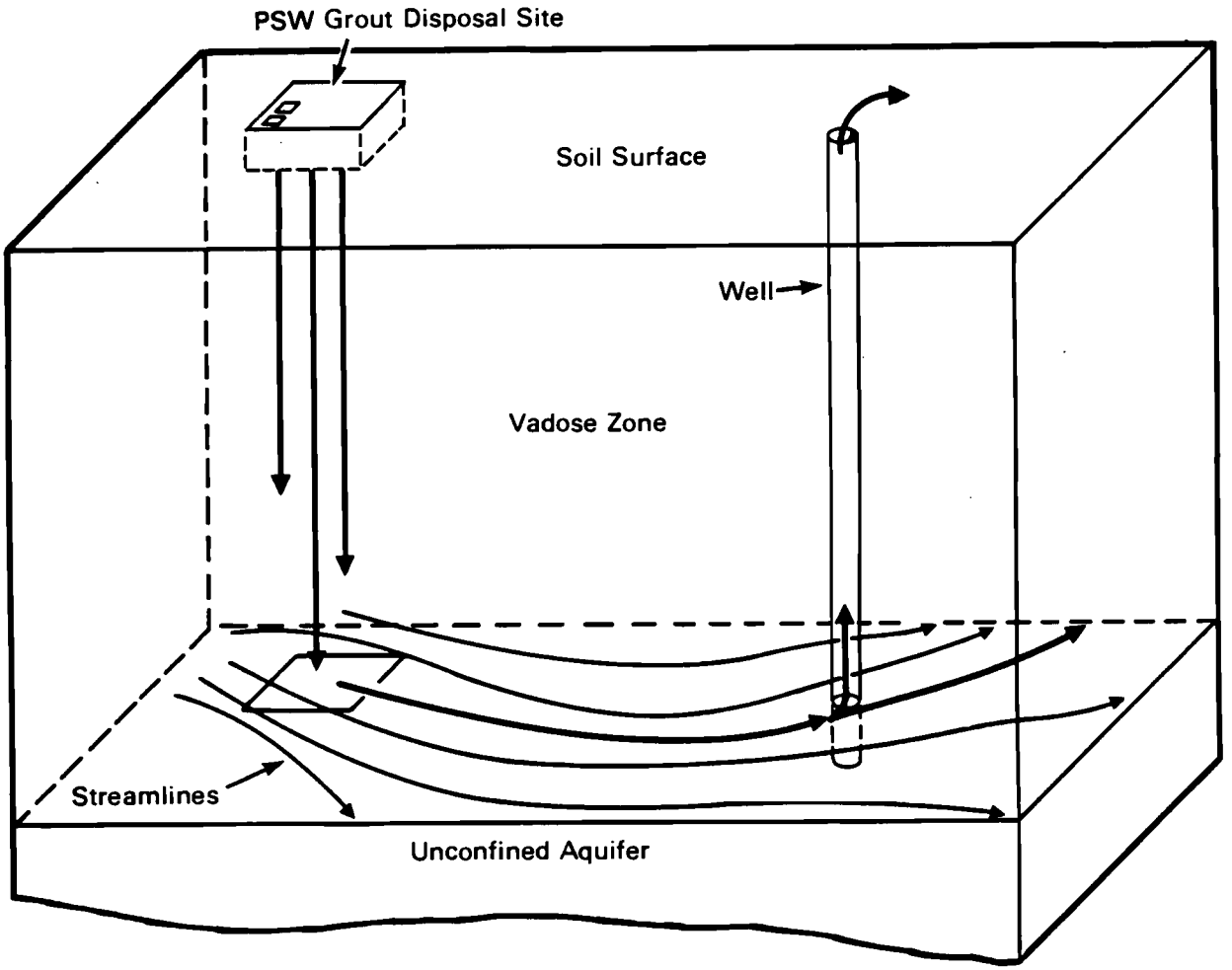


FIGURE 5.1. Conceptual Model for Contaminant Migration

As a basis for evaluating long-term performance, concentrations of regulated nonradioactive chemical species are compared to drinking water standards in Section 5.4 at both the groundwater well and in the Columbia River. To measure the long-term radiological impacts, exposure to the public is calculated based on ingesting the contaminated water from the hypothetical well, either through drinking the water directly from the well, or through eating crops and animals that were irrigated/fed with the contaminated water. Additionally, exposure is calculated (Section 6.3) as a result of drinking or using hypothetically contaminated Columbia River water.

5.2 MIGRATION THROUGH THE VADOSE ZONE

Water flow and contaminant transport through the vadose zone are simulated by two models that are separate but linked. The details and assumptions of each are discussed separately.

5.2.1 Water Flow Through the Vadose Zone

This section describes the vadose zone model and explains how values were calculated for the model parameters (e.g., physical properties of the vadose zone and the travel time of water through the zone).

The vadose zone below the future grout site is assumed to be composed of several horizontal layers of different thicknesses that have abrupt interfaces. Preliminary investigations of the vadose zone near the future grout site (observations of stratified soil columns) indicate that a vertical variation in soil properties does exist, and a number of distinct regions were identified. Figure 5.2 illustrates this conceptual model of the vadose zone. The properties of each layer (e.g., water content, hydraulic conductivity as a function of water content, sorption distribution coefficient) are assumed to be homogeneous throughout each layer. However, different layers may have different properties.

Water movement through the vadose zone is assumed to be steady-state, one-dimensional (vertically downward), and described by Darcy's law for unsaturated flow driven by a unit hydraulic gradient (i.e., other forces are assumed negligible compared to gravitational forces). Thus, the water flux (equal to the recharge rate for our considerations) is equal in magnitude to the hydraulic conductivity

$$q = K(\Theta) \qquad (5)$$

where q = magnitude of steady-state water flux (recharge rate)

Θ = water content

$K(\Theta)$ = hydraulic conductivity at water content Θ

Because the water flux is assumed constant throughout all layers of the vadose zone, it follows from Equation (5) that the hydraulic conductivity of

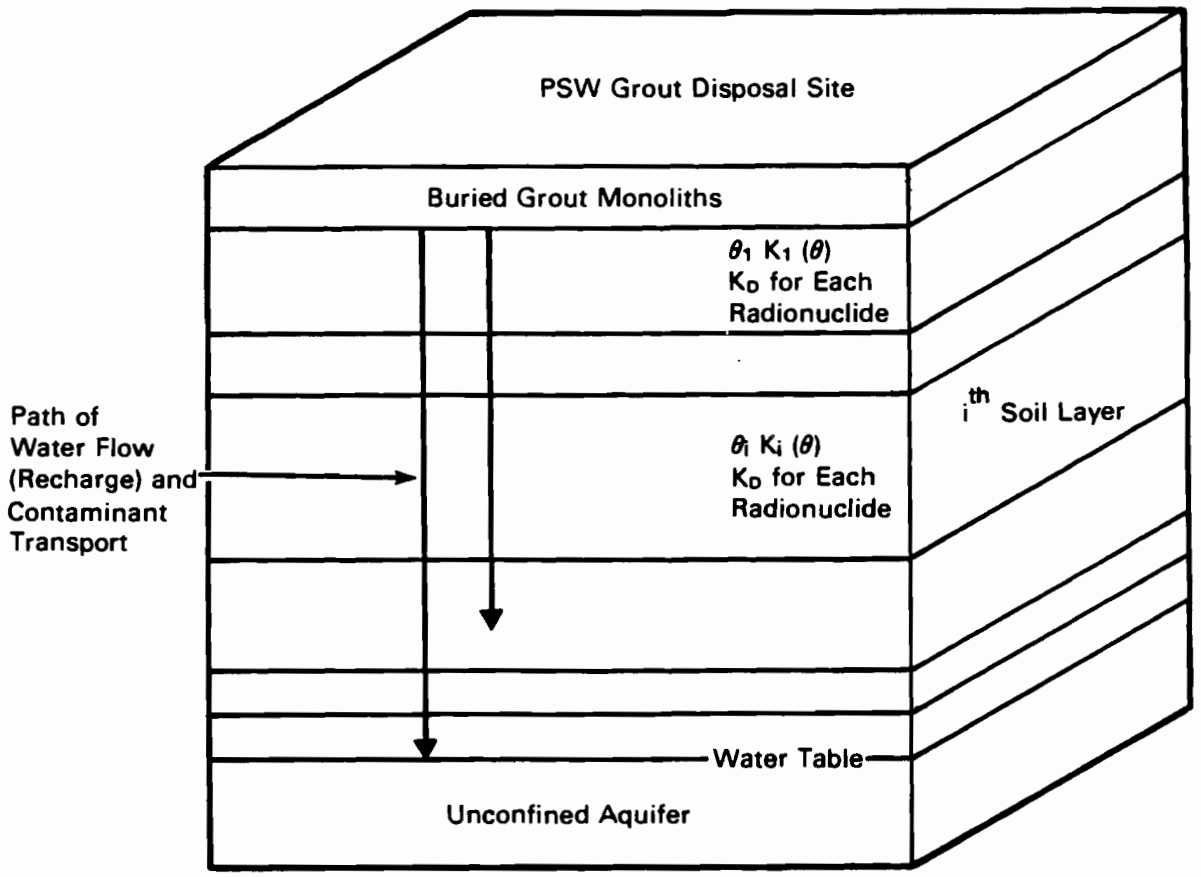


FIGURE 5.2. Conceptual Model of the Vadose Zone and Associated Water Flow and Contaminant Transport

all layers must be the same. If the layers have different hydraulic properties, the water content of a given layer must have a value (Θ_i for the i^{th} layer) such that

$$K_i(\Theta_i) = q \qquad (6)$$

where $K_i(\Theta)$ is the hydraulic conductivity function for the i^{th} soil layer.

The travel time of water through a given layer is calculated as the layer thickness divided by the average pore water velocity. The average pore water velocity is calculated as the water flux divided by the average water content.

The total travel time for water through the entire vadose zone is calculated by summing the travel times through individual layers:

$$t_w = \sum_{i=1}^{n} t_{wi} = \left(\sum_{i=1}^{n} \theta_i(q)\, T_i \right)/q \qquad (7)$$

where t_w = total travel time of water through vadose zone
t_{wi} = travel time of water through the i^{th} layer
n = number of soil layers
T_i = thickness of the i^{th} soil layer

and where the water content of the i^{th} layer has been written to show that it depends of the water flux, q. Unstable flow and viscous "fingering" caused by the abrupt layer interfaces or by preferential paths through the porous media are assumed not to occur.

The number of soil layers and the corresponding thicknesses used in the calculations of water travel time were taken from a hypothetical soil profile thought to be representative of the stratification at a location under the grout disposal site. This representative soil profile was generated by interpolating data (Fecht, Last and Marratt 1979) from soil profiles (extending to the water table) taken below the 216-A-8 and 216-A-37 cribs, which are located on the north and south sides of the 241-AP tank farm and the grout disposal site. Figure 5.3 is a schematic diagram of this stratified column.

Hydraulic conductivity functions (factors that affect the rate of water flow) for materials in these layers were not available. Instead, the hydraulic conductivity functions were estimated by assuming that they would be the same as those for materials with similar particle sizes and geologic soil types. Particle size distributions, saturated hydraulic conductivities, and water retention characteristics had been previously determined for six soil samples taken from a 15-m-deep excavation at the 241-AP tank farm at the Hanford Site. One sample was taken from each major horizon observed in this excavation. A textural description of these samples is given in Table 5.1. These data were used to obtain approximate hydraulic conductivity functions for model calculations.

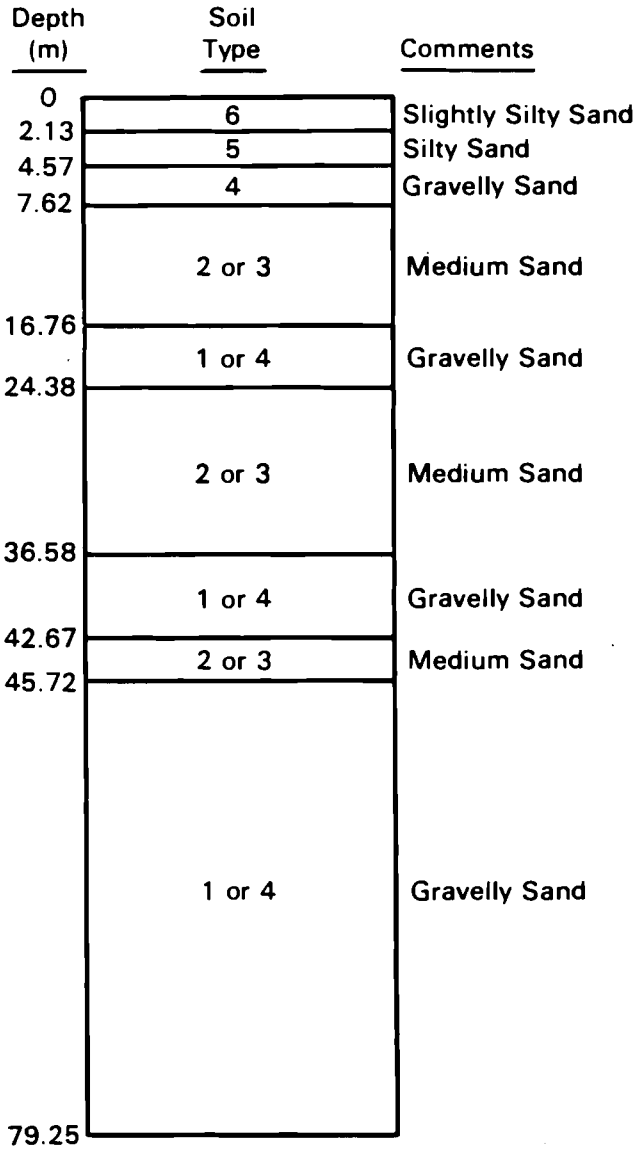


FIGURE 5.3. Schematic of Soil Profile Identifying Soil Types Using Available Data

Figure 5.3 shows which of the six 241-A soil samples were used to approximate the properties of each layer.

Table 5.2 presents the soil profile data used to generate Figure 5.3 along with the travel times to the unconfined aquifer through various combinations of soils. The numbers under columns A to D represent the combinations of soil types selected to represent each layer. By using predominantly

TABLE 5.1. Textural Description of Soil Samples Collected from 241-AP Tank Farm Excavation

Soil Sample Number and Type		Description
1	Sandy gravel	Unconsolidated, horizontally bedded, very coarse sand with very fine to very coarse pebbles
2	Sand	Well-consolidated, horizontally bedded, medium to coarse sand
3	Sand	Horizontal beds of well-consolidated fine to medium sand, to coarse sand
4	Gravelly sand	Very unconsolidated coarse sand with pebbles and small cobbles
5	Silty sand	Very well-consolidated horizontally bedded, very fine sand and silt
6	Slightly silty sand	Fine to very fine sand loess

coarse or fine media, one can establish bounds on the travel times associated with the uncertainty due to soil types. The data below 15 m were used to determine the travel time from the level of the proposed grout monoliths to the water table. The distance from the bottom of the monoliths to the water table was 64 meters. The travel times listed at the bottom of columns A through D are considered representative of the existing soil column. No major fine sand, silt, or clay layers were indicated above the water table in the stratigraphic cross sections beneath the 216-A-8 and 216-A-37 cribs in the 200-East Area. However, to show the effect of a fine sand layer (such as occurs at the surface), a 3.05-m thick layer was included in columns E and F. The resultant travel time of column E should be compared to that of column D, and the travel time of column F should be compared to column A. These results indicate that for a 5.0 cm/yr steady-state recharge rate, minimum and maximum travel times are 99 and 143 years, respectively. These are the lower and upper bounds of the travel times for these soil data; other soil combinations will result in travel times that fall between these times. An infiltration rate of 0.5 cm/yr results in minimum and maximum travel times of 784 and

TABLE 5.2. Soil Profiles to the Water Table and Travel Times Through a 64-m Depth

Depth from Top to Bottom of Layer, m	Thickness of Layer, m	Soil Assignments					
		A	B	C	D	E	F
15.25 to 16.76	1.51	3	2	3	2	2	3
16.76 to 24.38	7.62	1	1	4	4	4	1
24.38 to 36.58	12.20	3	2	3	2	2	3
36.58 to 42.67	6.09	1	1	4	4	4	1
42.67 to 45.72	3.05	3	2	3	2	6[a]	6[a]
45.72 to 79.25 (water table)	33.53	1	1	4	4	4	1
Travel time in years							
for q = 5.0 cm/yr		99	118	124	143	149	109
for q = 0.5 cm/yr		784	936	957	1110	1166	862

(a) Substituted fine layer.

1,110 years, respectively. For calculations in this report, 925 and 100 years were selected as the travel times to be used in the transport model for the 0.5 and 5.0 cm/yr recharge rates, respectively.

5.2.2 Contaminant Transport Through the Vadose Zone

Contaminant transport through the vadose zone was modeled using the TRANSS solute transport code (Simmons, Kincaid and Reisenauer 1986). This code models one-dimensional, vertical transport of contaminants by an analytical solution of the convection-dispersion equation, including first-order radioactive decay and linear equilibrium sorption, and assuming a local-scale dispersion coefficient.

The remainder of this section describes the process for determining the quantity of each radionuclide that is sorbed within the layers of the vadose zone, and consequently, how long it takes sorbed radionuclides to travel through the vadose zone.

The linear equilibrium sorption coefficient (distribution coefficient), K_d, for a given radionuclide is assumed to be constant in all soil layers and for any water chemistry and flow rate. Specifically, this implies that when

several radionuclides are migrating simultaneously, the sorption of any one of them is still governed by its individual K_d value (i.e., there is no competitive sorption). Because the K_d value for a specific radionuclide depends on the chemical species in which the radionuclide occurs, this implies that the radionuclide exists in only one form.

Numerical values of K_d for each radionuclide, with the exception of strontium-90, were taken from a study related to single-shell tank contents (Delegard and Barney 1983). The composition of the liquid waste containing the radionuclides included organic complexants, and so the K_d values calculated from this study would be different than the K_d values for the radionuclides leaching from grout along with the primarily aqueous PSW constituents. Because the presence of organic complexants should act to reduce K_d values, it is conservative to use the values from the single-shell tank study. In addition, the salt content of single-shell tank liquid is much higher than that in the grout leachate, and higher salt content often translates into a lower K_d. Because of the lack of specific data on sorption of non-radioactive hazardous chemicals, K_d values for all non-radioactive hazardous chemicals were assumed to be zero.

The K_d value for strontium-90 was experimentally determined by leaching simulated PSW grout and running a batch K_d test with leachates that contained strontium-90. The values measured were approximately 91 and 31 (Serne et al. 1986). In this PA, a K_d of 31 was used. There is no effective difference between using a K_d of 31 or 91, because with either K_d value, the strontium-90 decays to an insignificant quantity while traveling in the vadose zone.

Sorption tends to retard the movement of solution-phase chemicals in time relative to the travel time of water (t_w). In the assumed theory, mean travel times for sorbed chemicals can be related to the water travel time, t_w, by

$$t_c = R\, t_w = (1 + \rho_b K_d/\theta)\, t_w = [1 + \rho_p(1 - \theta_s)K_d/\theta]\, t_w \qquad (8)$$

where t_c = travel time of the contaminant
R = retardation factor
ρ_b = bulk density of the porous material

ρ_p = average particle density of the soil solids
Θ_s = saturated water content (equal to the porosity)

As stated above, the K_d value for a given contaminant is assumed constant for all soil layers; however, because the bulk density and water content differ between layers, the retardation factor will be different for each layer. Because using a different retardation factor for each layer would complicate the calculations (and because there is disagreement on the proper water content value to use in calculations of advective transport in unsaturated soils), a conservative constant value of R was used. In this case, the conservative approximation is the one that results in the fastest movement of the contaminant through the vadose zone; namely, the approximation that results in the smallest value of R. According to the determination of R given in Equation (8), this smallest value of R is achieved if the largest value of water content is used, i.e., the saturated water content.

The mean travel time of a radionuclide through one layer (the i^{th} layer), t_{rni}, is given by $t_{rni} = R_i t_{wi} = [1 + \rho_b K_d/\Theta_i] t_{wi}$, according to Equation (8). Furthermore, the mean total travel time of a radionuclide through the vadose zone, t_{rn}, is given by

$$t_{rn} = \sum_{i=1}^{n} R_i t_{wi} = \sum_{i=1}^{n} (1 + \rho_b K_d/\Theta_i)\, t_{wi} \qquad (9)$$

If, as argued above, we assume that $\Theta_i = \Theta_s$ for all layers, R_i becomes a constant for all layers and we have

$$t_{rn} = R \sum_{i=1}^{n} t_{wi} = R t_w = (1 + \rho_b K_d/\Theta_s)\, t_w \qquad (10)$$

Here, t_{rn} represents the conservative approximation of the mean total travel time of the radionuclide. Because K_d has been assumed to be zero for nonradioactive hazardous chemicals, R_i would equal one, and the mean total travel time for these chemicals would just be equal to t_w. Note that the travel times as calculated are mean travel times, not actual ones. Several phenomena

(e.g., dispersion) will create a range in actual travel times of contaminant molecules moving through the vadose zone, rather than a single constant travel time for all molecules.

Table 5.3 presents half-lives, K_d values, retardation factors, and vadose zone radionuclide travel times for each significant radionuclide and for two recharge scenarios. The climate of the site, soil characteristics, and vegetation affect the rate at which meteoric water percolates deep enough into the vadose zone to become recharge to the aquifer and carry contaminants to the unconfined aquifer. To quantify the effect of the groundwater recharge rate on the performance of the grout disposal system, two average annual recharge rates were analyzed. Values of 0.5 and 5.0 cm/yr were chosen to "bridge" expected recharge rates under drier and wetter conditions at the disposal site. The lower end of this range represents recharge rates considered to be typical of present-day dry climatic conditions (15 cm/yr precipitation), while the upper end represents a hypothetical wetter climate (30 cm/yr precipitation) (Kirkham and Gee 1983).

TABLE 5.3. Radionuclide Travel Times in the Vadose Zone for Recharge Rates of 0.5 and 5.0 cm/yr

Nuclide	Half-life, yr	Distribution Coefficient (K_d)	Retardation[a] Factor (R)	Radionuclide Travel Time, yr	
				0.5 cm/yr Recharge	5.0 cm/yr Recharge
Carbon-14	5.7×10^3	0	1	925	100
Strontium-90	2.8×10^1	31[b]	170	157,000	17,000
Technetium-99	2.1×10^5	0	1	925	100
Iodine-129	1.6×10^7	0	1	925	100
Uranium-238	4.5×10^9	0	1	925	100
Plutonium-239	2.4×10^4	21[c]	115	106,000	11,500
Americium-241	4.3×10^2	5.6[c]	31.5	29,000	3,150

(a) Assuming $\theta_s = 0.33$ and $\rho_b = 1.8$ g/cm^3.
(b) Serne et al. (1986).
(c) Delegard and Barney (1983).

5.3 MIGRATION THROUGH THE UNCONFINED AQUIFER

As with the vadose zone, water flow and contaminant transport through the unconfined aquifer were simulated by two models that are separate but linked. The details and assumptions of each are discussed separately.

5.3.1 Water Flow Through the Unconfined Aquifer

Water flow through the unconfined aquifer was modeled using the VTT groundwater flow code. Details of the theory, assumptions, and limitations of this code are given in the draft environmental impact statement (EIS) for the disposal of Hanford defense waste (U.S. DOE 1986b), as well as in the code documentation report (Reisenauer 1979a,b,c).

The code determines kinematic pathlines and streamtubes for two-dimensional, steady-state, vertically averaged water flow in an aquifer of variable thickness. The aquifer is also assumed to be isotropic (the hydraulic conductivity or transmissivity at a point does not depend on the direction of water flow at that point) and horizontally heterogeneous (with respect to saturated hydraulic conductivity or transmissivity). The water flow model in VTT is based on the Boussinesq equation for incompressible fluid flow through a saturated, rigid porous media with a free-surface boundary condition. Vertical velocities, flow in the capillary fringe, and seepage between aquifers are assumed negligible. The numerical solution algorithm uses a finite difference approximation to the differential equation of flow, and solves the resulting set of algebraic equations with a Newton iteration technique.

Figures 5.4 and 5.5 show groundwater contours and streamlines near the 200-Area waste sites, as calculated by VTT, using recharge rates of 0.5 and 5.0 cm/yr, respectively (U.S. DOE 1986b).

Obtaining Values for the Transmissivity Field

The model assumes a spatially varying transmissivity field that is piecewise constant with transmissivity values constant over a finite region associated with each node. Numerical values representing the transmissivity field, which are required as inputs to the model, were obtained by calibrating the model with data of water potential surface for the unconfined aquifer. These data were collected during periods of Hanford Site operations, and

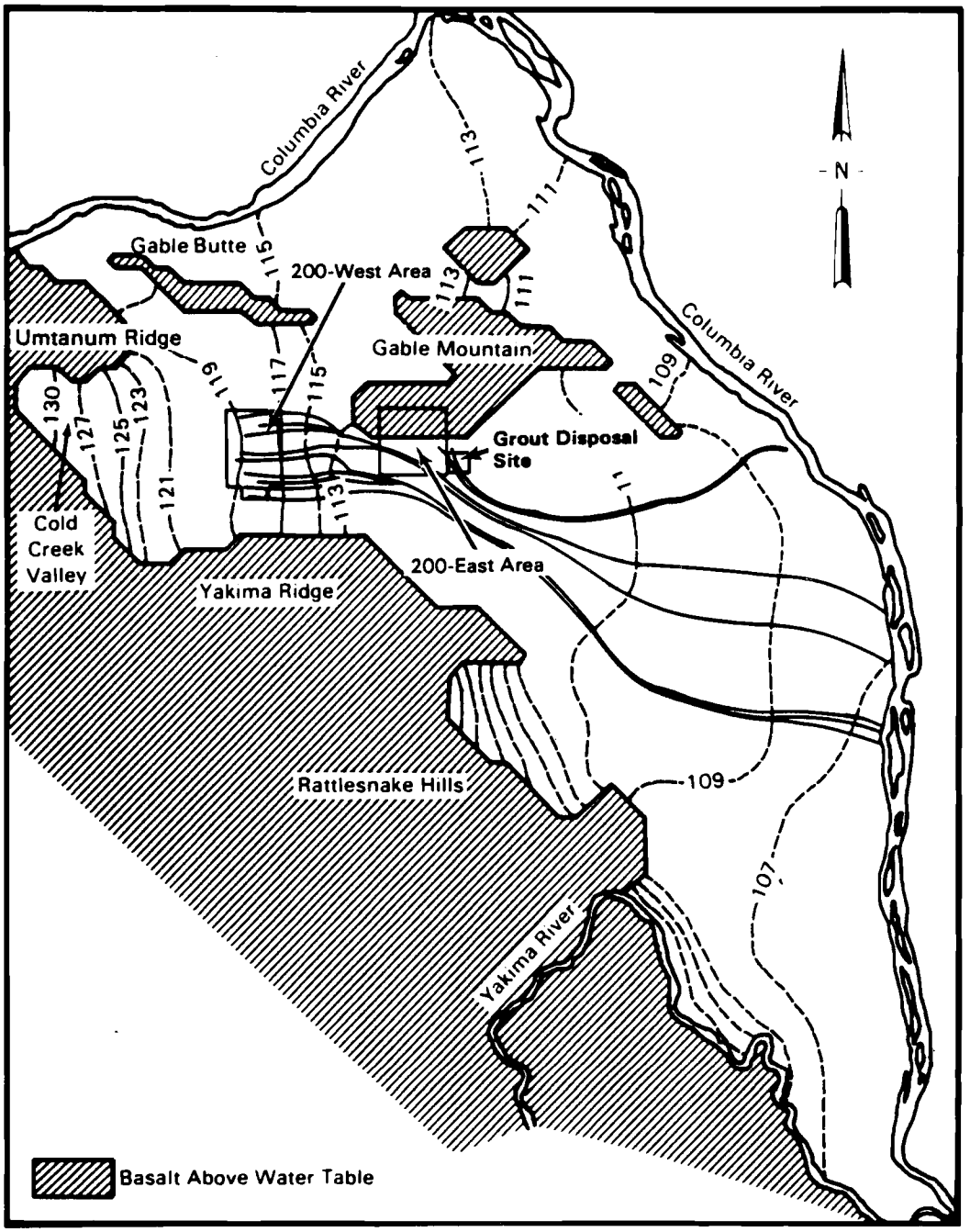


FIGURE 5.5. Simulated Groundwater Contours (in meters above mean sea level) and Streamlines from the 200-Area Plateau, 5.0 cm/yr Recharge Rate

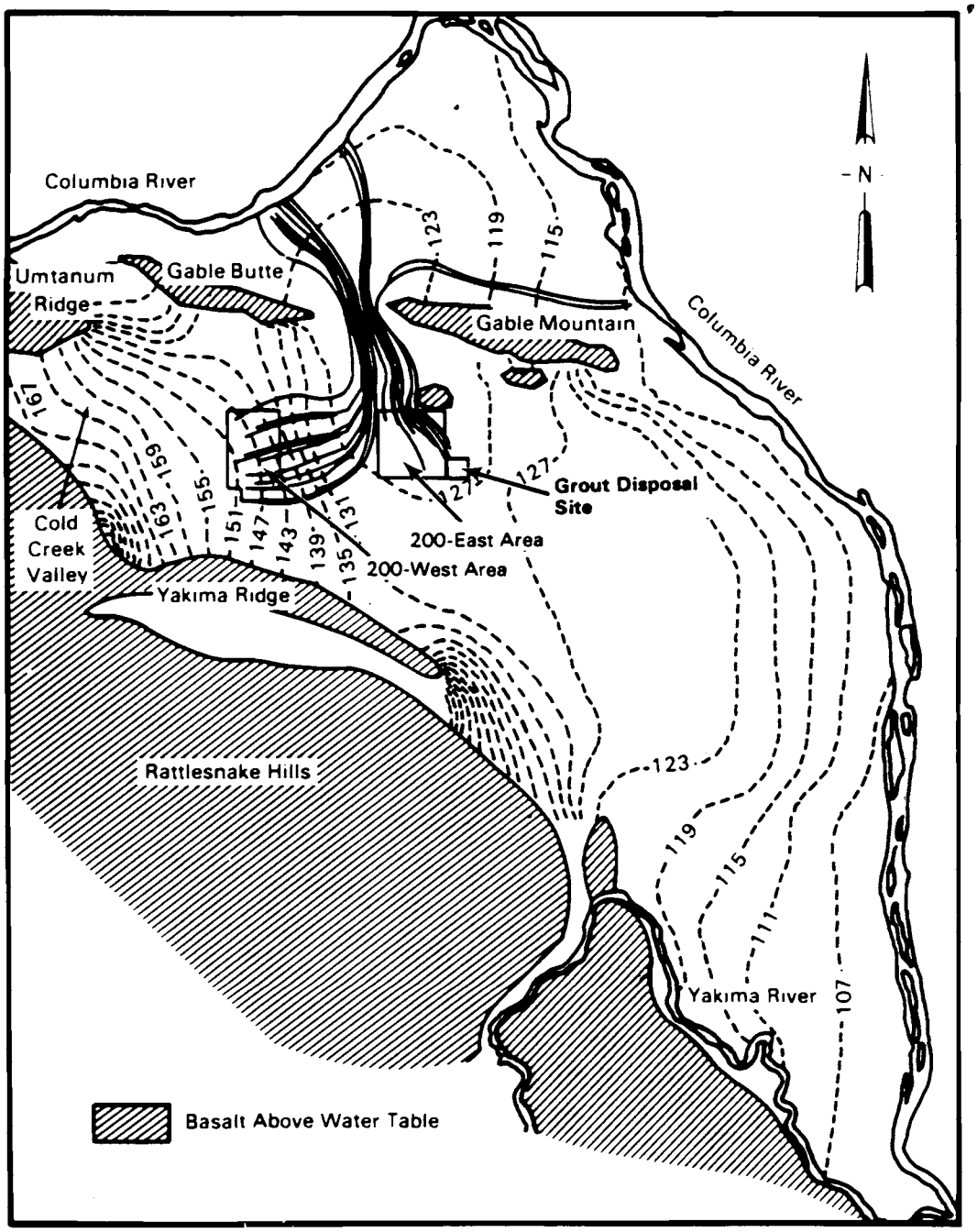


FIGURE 5.4. Simulated Groundwater Contours (in meters above mean sea level) and Streamlines from the 200-Area Plateau, 0.5 cm/yr Recharge Rate

therefore represent water table elevations that have been affected by various water infiltration and pumping activities associated with Hanford operations. Knowledge of the spatial distribution of unconfined aquifer thickness was used to obtain values of vertically averaged, saturated hydraulic conductivity from these calibrated transmissivity values. These resulting values of hydraulic conductivity were used in the VTT model for the simulations of the groundwater contours shown in Figures 5.4 and 5.5. In other words, each value of transmissivity required as input to VTT was obtained as the product of a value of the aforementioned hydraulic conductivity field and the not-yet-calculated thickness of the aquifer at that location. The simulations of the 0.5 and 5.0 cm/yr recharge scenarios have shown that the water table elevations (and hence, the aquifer thicknesses) predicted to exist in these scenarios are not substantially different than the measured elevations that were used for calibration. For this reason, it is assumed that only slight error may exist in predictions of future groundwater flow characteristics (i.e., when the water infiltration and pumping activities associated with Hanford operations have ceased) when using properties obtained by calibration with data taken during periods of Hanford activity.

Calculating the Annual Dilution Volume

Contaminants entering the unconfined aquifer directly below the PSW grout disposal site are assumed to mix with the flowing groundwater. The concentration of contaminants flowing in the groundwater is a function of the rate at which the groundwater flows beneath the disposal site. The greater the volume of water into which the released contaminants are mixed, the lower their concentrations in the groundwater.

There are two hypothetical cases to be considered when calculating the ultimate dilution volume in the unconfined aquifer. In one case, the amount of dilution is calculated as the product of four parameters: 1) the stream-tube intercept width, 2) an assumed mixing depth, 3) the porosity of the aquifer, and 4) the groundwater velocity. These parameters are discussed in detail in the following subsections. The resulting value is used as the annual dilution volume in cases in which the downgradient user of groundwater is assumed to pump less water per year than this dilution volume. In this

instance, the well is only drawing part of the contaminated water that is passing by, and the conservative assumption is that all of the water drawn is contaminated at a concentration governed by the dilution volume in the aquifer.

The second case occurs when more water is used than the aquifer dilution volume described above. In this case, the conservative assumption is that the well draws in all of the contaminated water passing by. However, to supply the needs of the user, it must also draw in uncontaminated water as well. Therefore, the concentrations of contaminants in the water used are further diluted by the extra uncontaminated water pumped.

Streamtube Intercept Width. Figure 5.1 can be referred to again to illustrate the transport scenario for the unconfined aquifer that is used to calculate the volume of water into which a yearly release of a contaminant mixes. The streamtube intercept is the dimension of the vertical projection of the grout site that is perpendicular to the water flow direction. The smaller the streamtube intercept width, the less dilution occurs. Hence, it is conservative to assume that the intercept is equal to the smallest plan dimension of the grout disposal site. The smallest plan dimension of the PSW grout disposal site, 50 m, is therefore used in the calculations.

Mixing Depth. By the time a contaminant is drawn up into a well, it has mixed downward through a certain depth below the water table. The VTT model is based on a vertically averaged set of equations, hence it does not calculate vertical flow. It is assumed that any well beyond the PSW grout disposal site that may take up water will penetrate the unconfined aquifer with a screened casing extending from the water table to 5 m below it. If the true mixing depth is less than 5 m, uncontaminated water will simultaneously be drawn into the well, and mixing will occur in the well in a manner such that the resulting radionuclide concentration is the same as that calculated by assuming 5 m for the mixing depth. If the true mixing depth is greater than 5 m, the radionuclide is already more dilute at the well than with a 5-m assumed mixing depth, and so this assumption is conservative. The present calculation could be less than conservative if the actual future well had a screen of less than 5 m in length and if the true mixing depth were less than

5 m. However, the 5-m mixing depth was chosen because it is the minimum depth of penetration that would accommodate the submergence necessary for operation of pumps using current technology.

Aquifer Porosity and Groundwater Velocity. The product of the streamtube intercept and the mixing depth gives the cross-sectional area of the aquifer that is perpendicular to the direction of flow (50 m x 5 m = 250 m^2). Multiplying this by two factors--the annual distance of flow of water passing through this area (which is equal to the groundwater velocity when expressed on a per-year basis) and the effective porosity of the soil in this area--gives the annual dilution volume. The velocity of the water passing under the site is assumed to be an average of the velocities across the streamtube, as calculated by the VTT code. This value is 182 or 230 m/yr for recharge rates of 0.5 or 5.0 cm/yr, respectively (U.S. DOE 1986b). To be consistent with past modeling, the effective porosity of the Hanford unconfined aquifer was assumed to be 0.1 (Bierschenck 1959).

Annual Dilution Volume: The Resulting Values. Using the parameter values described above, the annual dilution volume would be 4.6 x 10^3 cubic meters at the recharge rate of 0.5 cm/yr, and 5.8 x 10^3 cubic meters at the recharge rate of 5.0 cm/yr for the case where this volume is greater than that required each year by a downgradient user.

However, the value for the annual dilution volume must be selected to be compatible with the groundwater exposure scenarios. At present, the radiological exposure model assumes that the downgradient user farms a 20,000 m^2 parcel of land with an irrigation rate of 150 L/m^2-month during a 6-month-per-year irrigation period (Napier, Peloquin and Strenge 1986). The groundwater-use exposure scenario requires an annual volume of 1.8 x 10^4 cubic meters per year, which is greater than both values calculated above. Hence, 1.8 x 10^4 cubic meters was used as the annual dilution volume for the radiological exposure analysis presented in Chapter 6.0.

5.3.2 Contaminant Transport Through the Unconfined Aquifer

The VTT water flow model generates pathlines that can be used to represent two-dimensional advective solute transport in the unconfined aquifer.

(For steady-state conditions, these pathlines are the same as the streamlines of water flow.) Contaminants are assumed to enter the unconfined aquifer through an area of the water table that is the vertical projection of the PSW grout field onto the saturated flow field. The contaminants are subsequently transported by the groundwater flowing between the pathlines that enclose the projected area. Figure 5.6 illustrates this scenario. A leached contaminant is assumed to remain entirely within the streamtube bounded by these two pathlines, i.e., no transverse dispersion of radionuclides is assumed. The width of the streamtube is determined by the dimension of the vertical projection of the grout field that is perpendicular to the flow direction. Longitudinal dispersion is assumed to occur solely because of variations in advective velocities and pathlengths associated with pathlines within the streamtube.

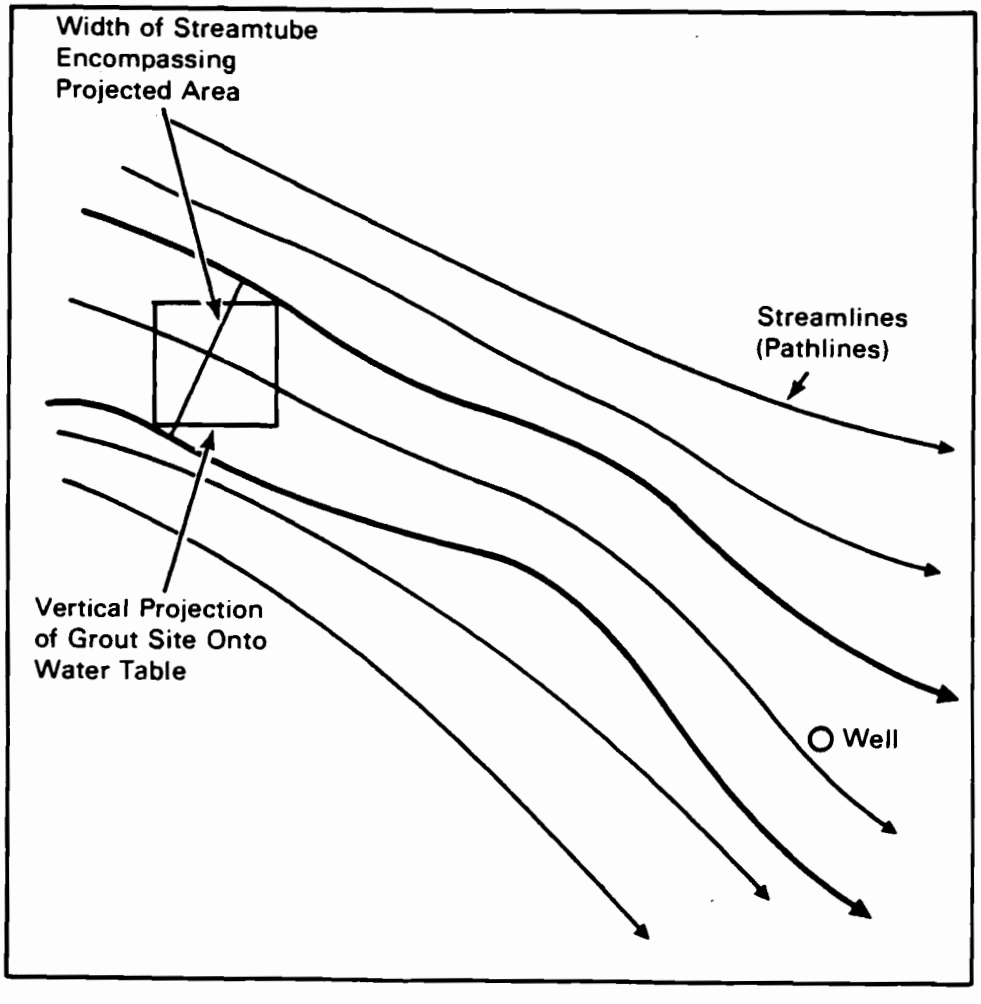


FIGURE 5.6. Conceptual Model for Contaminant Transport in the Unconfined Aquifer Along Water Flow Pathlines

The TRANSS computer code, which was used to model contaminant transport to the vadose zone, was also used to model contaminant transport along the one-dimensional streamtube in the unconfined aquifer. Radioactive decay and sorption are accounted for, and the same K_d values (one for each radionuclide) that TRANSS used for the vadose zone model are used in the unconfined aquifer. The associated assumptions are also the same.

5.4 NONRADIOLOGICAL RESULTS

The long-term ability of the grout disposal system to contain nonradioactive regulated chemicals can be measured by direct comparison to concentration standards at two locations: 1) the hypothetical well located 5 km downgradient from the grout monoliths, and 2) the Columbia River. Chemical concentrations in the well and river, based on the projected inventories and the release and transport assumptions described above, are listed in Table 5.4. The values in Table 5.4 represent incremental increases, which should be added to the existing level of contamination from other sources to obtain the absolute value of the actual contaminant concentration.

As previously mentioned, although the Hanford groundwater does not constitute a public water supply, drinking water limits are listed in Table 5.4 for comparison. All calculated incremental concentrations are below the limits at both the well and the river. The Hanford Reach of the Columbia River is governed by Class A water quality standards for the State of Washington (WAC 1984). These standards do not list specific limits for these inorganic chemicals, but they do include limits on biological waste, turbidity, thermal waste (heat), and aesthetic qualities of the river. The incremental increases of chemicals postulated to occur in Columbia River water as a result of disposed PSW are very low, and will comply with all Class A water quality standards.

Radionuclide concentrations must be translated to resulting doses before being compared to regulatory limits. Therefore the radionuclide concentrations are not listed here, but are used as input for the groundwater exposure scenarios described in Chapter 6.0. The resulting doses are provided in Chapter 6.0, Section 6.3.

TABLE 5.4. Calculated Increase in Concentrations of Regulated Chemicals in a Hypothetical 5-km Well and in the Columbia River, mg/L

	In 5-km Well Water		In Columbia River Water		Washington State Drinking Water Limit[a]
	Recharge Rate, cm/yr				
	0.5	5.0	0.5	5.0	
Primary Contaminants[a]					
Arsenic	2×10^{-2}	1×10^{-2}	7×10^{-10}	7×10^{-10}	0.05
Barium	6×10^{-4}	4×10^{-3}	3×10^{-11}	2×10^{-10}	1.0
Cadmium	3×10^{-5}	2×10^{-4}	1×10^{-12}	1×10^{-11}	0.01
Chromium	1×10^{-3}	1×10^{-2}	4×10^{-11}	6×10^{-10}	0.05
Fluoride	3×10^{-2}	2×10^{-1}	1×10^{-9}	1×10^{-8}	2.0
Lead	4×10^{-3}	3×10^{-2}	2×10^{-10}	2×10^{-9}	0.05
Mercury	1×10^{-4}	9×10^{-5}	5×10^{-12}	5×10^{-12}	0.002
Nitrogen	3×10^{-1}	3×10^{-1}	1×10^{-8}	1×10^{-8}	10.0
Selenium	1×10^{-3}	8×10^{-4}	5×10^{-11}	5×10^{-12}	0.01
Silver	1×10^{-3}	8×10^{-3}	4×10^{-11}	5×10^{-10}	0.05
Secondary Contaminants[a]					
Chloride	3×10^{-1}	3×10^{-1}	2×10^{-8}	2×10^{-8}	250
Copper	8×10^{-4}	7×10^{-3}	4×10^{-11}	4×10^{-10}	1.0
Iron	3×10^{-4}	3×10^{-3}	1×10^{-11}	2×10^{-10}	0.3
Manganese	3×10^{-4}	2×10^{-3}	1×10^{-11}	1×10^{-10}	0.05
Sulfate	3×10^{1}	2×10^{1}	1×10^{-6}	1×10^{-6}	250
Zinc	3×10^{-4}	2×10^{-3}	1×10^{-11}	1×10^{-10}	5.0

(a) Per Washington State standards for public water supplies (WAC 1985).

6.0 RADIOLOGICAL ASSESSMENT

This chapter presents the long-term radiological impacts from the disposal of grouted PSW. Radiation doses are calculated for members of the general public who receive exposure by either directly contacting the grouted waste form or by being exposed to radionuclides that have migrated from the grout into the groundwater. Calculated doses represent incremental increases. Both natural and human-induced events that could lead to contact with the disposed radionuclides have been addressed. The dosimetry models, pathway models, calculational method, and the resulting calculated impacts are presented below.

6.1 POSTULATED EVENTS

Table 6.1 lists events that past PAs have postulated to lead to significant releases of radionuclides from waste disposed near the ground's surface at Hanford.

TABLE 6.1. Postulated Events Leading to Radiological Impacts[a]

- Residential Home Garden
- Drilling
  - Resource Exploration
  - Water well
- Post-drilling Habitation
- Excavation
- Climate State
  - Present
  - Drier
  - Wetter
- Irrigation
  - Onsite
  - Offsite
- Contaminated Water Supply
- Glacial Flooding
- Other Surface Flooding
  - 100-Year Flood
  - Dam Failure
- Wind Erosion
- Magmatic Activity
- Seismic Activity
- Criticality
- Terrorism
- Warfare

(a) Source: U.S. DOE 1986(b).

Of these events, eight were judged to have sufficient probability and/or consequence for grout disposal to warrant further detailed analysis:

- contaminated water supply
- irrigation
- changing climate
- drilling
- excavation
- residential garden
- post-drilling habitation
- post-excavation habitation.

A suite of exposure scenarios was established based on assumptions specific to the analysis of long-term performance of the PSW grout system. The radiological impacts associated with these scenarios are presented in Section 6.3 (Groundwater Impacts) and Section 6.4 (Intruder Impacts). Methods for calculating radiation doses are presented in Section 6.2.

6.2 CALCULATIONAL METHODS

This section describes the dose model used, then discusses the methods for calculating groundwater doses and doses to intruders.

6.2.1 Dosimetry

The dose model used in this report is derived from that originally endorsed by the International Commission on Radiological Protection (ICRP 1959) in Publication 2 for body burden and maximum permissible concentration. Effective decay energies for radionuclides are calculated using the ICRP model. This model is based on the assumption that the entire quantity of a given radionuclide is located at the center of a spherical organ with an appropriate effective radius (Soldat 1976). Metabolic parameters for the standard man are used (ICRP 1975); some of the parameters are updated from later ICRP publications.

Several radionuclides are handled as special cases. For the radionuclides tritium and carbon-14, the accumulated dose for the organs, total body,

and bone are calculated as described above. Because these radionuclides distribute evenly in the rest of the body, the doses for all the other organs are set equal to that for total body.

The model for the gastrointestinal (GI) tract is as follows. The GI tract--stomach, small intestine (SI), upper large intestine (ULI), and lower large intestine (LLI)--is modeled as a four-compartment system with a plug flow. Because no long-term storage or retention of radionuclides occurs in the GI tract, the dose to the GI tract in any one year is equal to the dose commitment for that year. The portions of the GI tract are assumed to be irradiated by radionuclides that are uniformly distributed in the material passing through each compartment.

The internal distribution of radionuclides following inhalation adds a degree of complexity because of radionuclide retention in the lungs. The model of the respiratory tract adopted by the ICRP Task Group on Lung Dynamics (ICRP 1966) forms the general basis for the mathematical models developed to calculate the dose from inhalation of radionuclides.

A new dosimetry model has recently been developed and recommended by the ICRP in ICRP-26 (ICRP 1977) and applied in ICRP-30 (ICRP 1979). The new model is based on more recent human metabolic parameters and applies a more realistic approach for uptake and retention of radionuclides in body organs. For example, the contribution to organ dose resulting from decay of radionuclides in other organs ("crossfire") is now accounted for. Rather than report the individual organ doses, the concept of an "effective whole-body dose" (the sum of the product of each organ dose multiplied by its appropriate weighting factor) is used. The effective whole-body dose is then used for comparison to a stochastic dose limit. Stochastic in this context means that impact is proportional to dose; i.e., no threshold is assumed. The stochastic effective dose equivalent limit recommended for an individual in the general public, according to ICRP-26, is 500 mrem/yr. In addition, ICRP-26 states that when prolonged exposures are expected, the annual dose limit should be 100 mrem/yr.

As mentioned, the dosimetry used in this report is based on the ICRP-2 model. An environmental assessment code using the more recent ICRP 26/30

dosimetry model is being developed at PNL but is not yet available for use. However, to determine whether the calculated doses in our PA would have varied greatly if the ICRP-26/30 dose model were used, the dose results calculated for the drinking water scenario (Section 6.3.1) were compared using ICRP-2 and ICRP-26/30 methods. The ICRP 26/30 doses were estimated using hand calculations for this comparison.

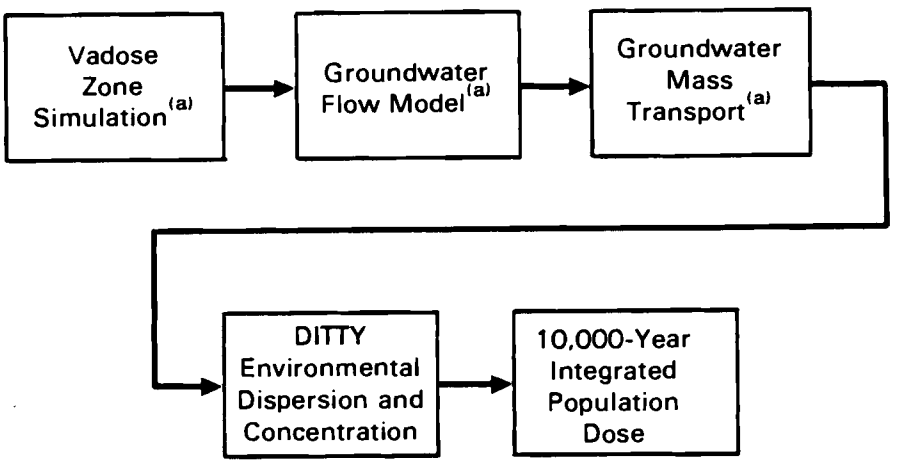
As shown in Table 6.2, the radiological impacts from the drinking water scenario calculated using ICRP-2 methods result in doses of 0.02 mrem/yr to the total body and 0.3 mrem/yr to the critical organ (bone). Using the newer ICRP 26 and 30 methodology, the calculated effective dose equivalent is 0.09 mrem/yr. Hence, the ICRP-26/30 dose is approximately three times below that of the ICRP-2 critical organ dose.

TABLE 6.2. Comparison of Radiation Dose for the Drinking Water Scenario Using ICRP-2 and ICRP-26/30 Methods, mrem/yr

Radionuclide	ICRP-2		ICRP-26/30
	Total Body	Bone	Effective Dose Equivalent
^{14}C	5.6×10^{-3}	2.7×10^{-2}	2.0×10^{-2}
^{238}U	1.4×10^{-2}	2.3×10^{-1}	7.0×10^{-2}
^{99}Tc	8.0×10^{-5}	2.1×10^{-4}	2.0×10^{-3}
^{129}I	2.0×10^{-7}	7.3×10^{-8}	6.2×10^{-6}
	2.0×10^{-2}	3.0×10^{-1}	9.0×10^{-2}

6.2.2 Method for Calculating Groundwater Doses

The computer program DITTY (Napier, Peloquin and Strenge 1986) estimates the time integral of collective dose over a period up to 10,000 years for time-variant radionuclide releases to surface waters, wells, or the atmosphere. The computer program was initially developed to determine the collective dose from high-level waste geologic repositories resulting from groundwater pathways, but other pathways are included as well. The relationship of DITTY to the hydrogeologic models described in Chapter 5.0 is shown in Figure 6.1.



(a) See Chapter 5.0 for details.

FIGURE 6.1. Use of Computer Models to Calculate 10,000-year Integrated Population Doses from Contaminant Release to Groundwater

Source terms for DITTY may be defined for releases to the atmosphere, to groundwater, to water wells, and to surface water via groundwater. The actual release rates are specified in an input file as the curies per year released for selected years following the start time of the calculation.

The DITTY code calculates a dose for any 10,000-year period. This period is broken into 143 periods of 70 years each. The average release during each period is calculated from the source-term data provided. The total-population dose to selected organs is determined for the population present in each period. The radioactivity present during any period is the sum of material uniformly released over that period and residual material in the environment from releases in previous periods. The dose is calculated for all contributing pathways of exposure, including external exposure, inhalation, and ingestion of contaminated water and foods.

6.2.3 Method for Calculating Intruder Doses

The ONSITE/MAXI computer program (Napier et al. 1984) is used to calculate a maximum annual dose to an individual from residual surface contamination. Exposure pathways that can be modeled include direct external

exposure to contaminated soil or building surfaces, inhalation of resuspended material, and ingestion of contaminated foods and aquatic products. The ONSITE/MAXI code calculates the time of the maximum dose rate to specific organs of reference and gives an annual dose for that organ. Special options are available to tailor the program to simulate a variety of decommissioned facilities such as reactors, low-level waste burial grounds, or other facilities for handling nuclear material.

As described in Section 6.4 of this document, the onsite scenarios are postulated to occur at a future date following the hypothetical loss of active institutional control of the Hanford Site. While warning markers, land-use records, and protective overburdens over the grout vaults should render intrusion events less likely, it is impossible to accurately predict human behavior over the long term; a determined individual can ignore, circumvent, or destroy any potential barrier. Thus, an individual could potentially receive a wide range of exposures depending on the magnitude of disruption of a site. Consequently, a suite of scenarios has been used that spans the range from negligible to significant site disruption.

The scenarios evaluated in this performance assessment include drilling, excavation, and residential gardening activities that hypothetically occur directly over the PSW grout disposal site. Drillers or diggers are assumed to penetrate areas of the highest radioactivity. The probability, or relative unlikelihood, of a scenario occurring is not factored into the calculations; the intrusions are assumed to occur and the results are presented.

For the scenarios involving farming, roots from all types of plants, including food crops, are assumed to penetrate to a depth of at least 5 m with a small percentage contacting the grouted wastes. Because the intrusion events are localized, the individuals receiving maximum doses are the intruders themselves. While the scenarios chosen for analysis represent a range of potential conditions, the parameters used for each scenario are selected to ensure conservatism.

6.3 GROUNDWATER IMPACTS

As described in Chapters 4.0 and 5.0, once the radionuclide contaminants are released from the grout vault, they migrate downward through the vadose zone. From the vadose zone they mix with the unconfined aquifer below the grout disposal site. Once in the aquifer, the radionuclides are transported away from the grout disposal site and may eventually reach the Columbia River, depending on their specific half-lives and adsorption characteristics.

Similar to the calculation of concentrations of regulated chemicals presented in Chapter 5.0, incremental concentrations of radionuclides in the unconfined aquifer and in the Columbia River were calculated as input for the dose assessment code for the groundwater exposure pathway. The time-dependent concentrations of key radionuclides in groundwater are shown in Figures 6.2 and 6.3 for recharge rates of 0.5 and 5.0 cm/yr, respectively.

To calculate radiological impacts, one must postulate locations at which humans could contact the contaminated groundwater. As mentioned, one assumed location is a hypothetical domestic well located between the PSW grout disposal site and the Columbia River. The location of this well was chosen so as to draw water from the maximally contaminated portion of the aquifer. The radiological impacts calculated for using water from this well are almost independent of distance from the PSW grout disposal site because conservative assumptions were made concerning the ability of contaminants to disperse in the aquifer. Hence a well located 10 km downgradient from the PSW grout disposal site is modeled as having essentially the same radionuclide concentrations as would a well located 100 m downgradient. Because the well scenarios described in the following sections are assumed to occur immediately after the loss of active institutional control, the well could conceivably be located anywhere from 0 km to 10 km from the disposal site. However, the most likely location for people to resettle many years into the future would be some distance from the 200-Area Plateau. Therefore, the well is assumed to be 5 km from the PSW grout disposal site.

Because contaminants may eventually reach the Columbia River, radiological impacts were also assessed downriver from the Hanford Site.

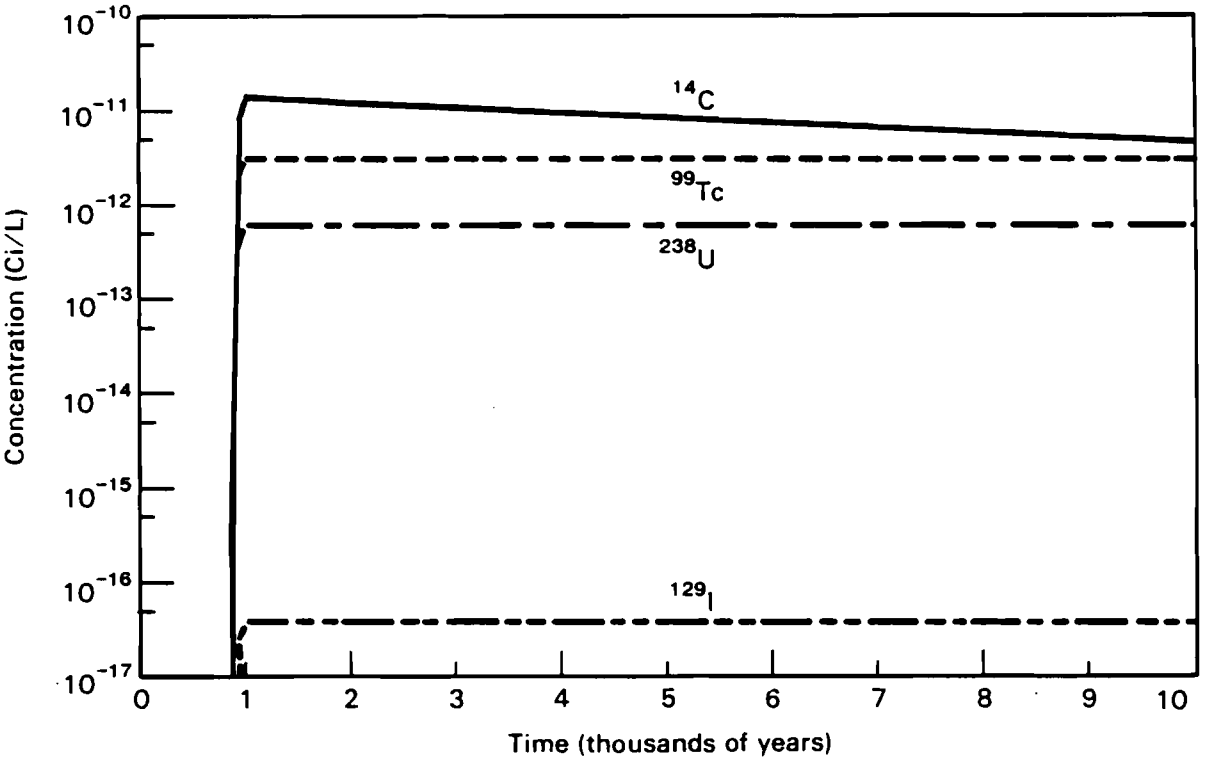


FIGURE 6.2. Concentrations of Key Radionuclides in Groundwater at 5-km Well, 0.5 cm/yr Recharge Rate

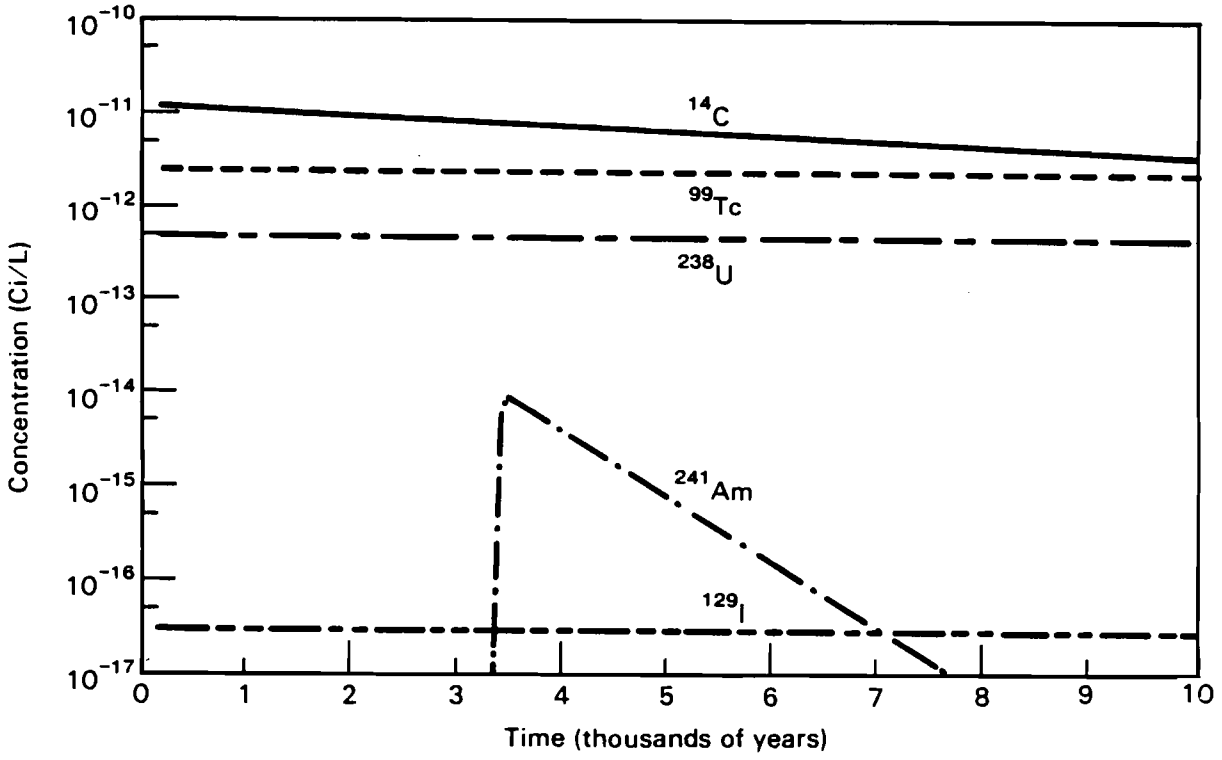


FIGURE 6.3. Concentrations of Key Radionuclides in Groundwater at 5-km Well, 5.0 cm/yr Recharge Rate

6.3.1 Drinking Well Water

A measure of the level of contamination of groundwater is the radiation dose caused by drinking the water. Annual and lifetime doses to an individual drinking water from the hypothetical 5-km well are given in Tables 6.3 and 6.4. The maximally exposed hypothetical individual evaluated was a person who consumes 2 liters of water per day, as defined by the Nuclear Regulatory Commission (U.S. NRC 1977). Total-body and critical-organ doses are presented in these two tables as a function of recharge rate. The time at which the maximum doses are projected to occur is shown to be dependent on the assumed recharge rate. The maximum doses occur 600 years and 1600 years following disposal for the 5.0 and the 0.5 cm/yr recharge rates, respectively. The doses presented are the maximum doses expected to occur within a 10,000-year period. The dominant radionuclides in this scenario are uranium-238 (which contributes 89% of the dose) and carbon-14.

Because leach rate data specific to uranium is unavailable, the release rate of uranium from within the grout monolith to the surrounding soils was modeled as being congruent with the diffusional release rates of mobile ions such as nitrite and sodium. This assumption may overestimate the release rate for uranium, because it may be solubility-controlled and slower than diffusional release. Additional laboratory studies are being conducted to quantitatively measure the leach rate of uranium-238 from PSW grout. Because uranium-238 controls the dose rate under the current set of assumptions, a reduction in release rate could significantly reduce the projected doses for this scenario.

6.3.2 Full-Garden Scenario

Contaminated well water could be used for irrigation and livestock water, as well as for human drinking water. A full-garden scenario was postulated wherein an individual grows a large percentage of his food using the well for irrigation, as might occur on a small, 2-ha (5-acre) family farm. In addition to drinking contaminated water, the individual is exposed to radionuclides deposited on the soil via irrigation water and also to radioactivity accumulated in crops and animal products. Dietary parameters for this scenario are

TABLE 6.3. Calculated Maximum Radiation Doses to Individuals from the Drinking Water Scenario, mrem (Annual Doses)

0.5 cm/yr Recharge Rate (maximum exposure 1600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	5.6×10^{-3}	28	2.7×10^{-2}	10	5.6×10^{-3}	99	5.6×10^{-3}	97	5.6×10^{-3}	16
Uranium-238+D	1.4×10^{-2}	71	2.3×10^{-1}	89	0	0	0	0	1.9×10^{-2}	53
Technetium-99	8.0×10^{-5}	<1	2.1×10^{-4}	<1	2.6×10^{-5}	<1	0	0	1.1×10^{-2}	31
Iodine-129	2.0×10^{-7}	<1	7.3×10^{-8}	<1	0	0	1.6×10^{-4}	3	1.1×10^{-8}	<1
TOTAL	2×10^{-2}		3×10^{-1}		6×10^{-3}		6×10^{-3}		4×10^{-2}	

5.0 cm/yr Recharge Rate (maximum exposure 600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	4.7×10^{-3}	30	2.3×10^{-2}	11	4.7×10^{-3}	100	4.7×10^{-3}	97	4.7×10^{-3}	17
Uranium-238+D	1.1×10^{-2}	70	1.8×10^{-1}	89	0	0	0	0	1.5×10^{-2}	50
Technetium-99	6.3×10^{-5}	<1	1.6×10^{-4}	<1	2.0×10^{-5}	<1	0	0	8.6×10^{-3}	30
Iodine-129	1.6×10^{-7}	<1	5.8×10^{-8}	<1	0	0	1.3×10^{-4}	3	8.8×10^{-9}	<1
TOTAL	2×10^{-2}		2×10^{-1}		5×10^{-3}		5×10^{-3}		3×10^{-2}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

TABLE 6.4. Calculated Maximum Doses to Individuals from the Drinking Water Scenario, mrem (Lifetime Doses)

0.5 cm/yr Recharge Rate (maximum exposure 1600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	3.9×10^{-1}	28	1.9×10^{0}	11	3.9×10^{-1}	100	3.9×10^{-1}	97	3.9×10^{-1}	16
Uranium-238+D	9.7×10^{-1}	71	1.6×10^{1}	89	0	0	0	0	1.3×10^{0}	53
Technetium-99	5.6×10^{-3}	<1	1.5×10^{-2}	<1	1.8×10^{-3}	<1	0	0	7.6×10^{-1}	31
Iodine-129	1.4×10^{-5}	<1	5.1×10^{-6}	<1	0	0	1.1×10^{-2}	3	7.8×10^{-7}	<1
TOTAL	1×10^{0}		2×10^{1}		4×10^{-1}		4×10^{-1}		2×10^{0}	

5.0 cm/yr Recharge Rate (maximum exposure 600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	3.3×10^{-1}	30	1.6×10^{0}	11	3.3×10^{-1}	100	3.3×10^{-1}	97	3.3×10^{-1}	17
Uranium-238+D	7.7×10^{-1}	70	1.3×10^{1}	89	0	0	0	0	1.0×10^{0}	50
Technetium-99	4.4×10^{-3}	<1	1.1×10^{-2}	<1	1.4×10^{-3}	<1	0	0	6.0×10^{-1}	30
Iodine-129	1.1×10^{-5}	<1	4.0×10^{-6}	<1	0	0	8.8×10^{-3}	3	6.1×10^{-7}	<1
TOTAL	1×10^{0}		2×10^{1}		3×10^{-1}		3×10^{-1}		2×10^{0}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

presented in Napier, Peloquin and Strenge (1986). In addition, other parameters were required to estimate the dose to this individual. For example, it was assumed that the individual spends 50% of his time exposed to contaminated soil. It was also assumed that irrigation occurs for 6 months of the year at a rate of 150 L/m^2/month. Soil-to-plant concentration ratios and meat concentration ratios are from Napier, Peloquin and Strenge (1986).

Radiation doses to individuals for this scenario were estimated for the same well-water concentrations as shown in Figures 6.2 and 6.3. Tables 6.5 and 6.6 show the calculated annual and lifetime doses, respectively. As with the drinking water scenario, the radionuclide that makes the major dose contribution (75%) is uranium-238.

6.3.3 Radionuclide Migration to the Columbia River

Radionuclides and other contaminants that are leached into the groundwater could eventually reach the Columbia River. The rate at which radionuclides enter the river depends on five factors: 1) the rate at which they enter the groundwater, 2) their radioactive decay, 3) their adsorption characteristics, 4) the linear flow rate of the aquifer, and 5) distance to the river. The highly mobile radionuclides (e.g., iodine-129 and technetium-99) could reach the Columbia River within a few hundred years after the initiation of waste leaching. The less mobile radionuclides, e.g., cesium-137 and strontium-90, were shown to decay before ever reaching the water table. The rate of radionuclide release to the river is shown in Figures 6.4 and 6.5, at recharge rates of 0.5 and 5.0 cm/yr, respectively. It is assumed that the flow rate of the river past Hanford is 120,000 cubic feet per second (U.S. DOE 1986b).

The Columbia River is currently used for drinking, irrigation, and recreation by many people living downstream of Hanford. These uses are assumed to increase in the future. Currently, only a small fraction of the river's flow below Hanford is used for irrigation and drinking. (Water for the large irrigation projects in the area is primarily obtained from the Columbia River upstream of Hanford.) Within 80 km of Hanford, 2,000 people are currently estimated to eat food grown using irrigation water from the Columbia River, 70,000 people drink water from the river, and about

TABLE 6.5. Calculated Maximum Radiation Doses to Individuals from the Full-Garden Scenario, mrem (Annual Doses)

0.5 cm/yr Recharge Rate (maximum exposure 1600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	1.1×10^{-2}	22	5.5×10^{-2}	16	1.1×10^{-2}	30	1.1×10^{-2}	33	1.1×10^{-2}	11
Uranium-238+D	3.9×10^{-2}	77	2.9×10^{-1}	84	2.6×10^{-2}	69	2.3×10^{-2}	66	4.5×10^{-2}	44
Technetium-99	3.3×10^{-4}	1	8.6×10^{-4}	<1	1.1×10^{-4}	<1	2.9×10^{-14}	<1	4.5×10^{-2}	45
Iodine-129	6.9×10^{-7}	<1	4.0×10^{-7}	<1	2.4×10^{-7}	<1	3.5×10^{-4}	1	2.6×10^{-7}	<1
TOTAL	5×10^{-2}		3×10^{-1}		4×10^{-2}		3×10^{-2}		1×10^{-1}	

5.0 cm/yr Recharge Rate (maximum exposure 600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	1.3×10^{-2}	25	6.2×10^{-2}	18	1.3×10^{-2}	33	1.3×10^{-2}	36	1.3×10^{-2}	12
Uranium-238+D	3.8×10^{-2}	75	2.9×10^{-1}	82	2.5×10^{-2}	66	2.2×10^{-2}	63	4.4×10^{-2}	43
Technetium-99	3.3×10^{-4}	<1	8.7×10^{-4}	<1	1.1×10^{-4}	<1	3.0×10^{-14}	<1	4.5×10^{-2}	45
Iodine-129	6.9×10^{-7}	<1	4.0×10^{-7}	<1	2.4×10^{-7}	<1	3.5×10^{-4}	1	2.6×10^{-7}	<1
TOTAL	5×10^{-2}		4×10^{-1}		4×10^{-2}		4×10^{-2}		1×10^{-1}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

TABLE 6.6. Calculated Maximum Radiation Doses to Individuals from the Full-Garden Scenario, mrem (Lifetime Doses)

0.5 cm/yr Recharge Rate (maximum exposure 1600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	7.9×10^{-1}	22	3.9×10^{0}	16	7.9×10^{-1}	30	7.9×10^{-1}	33	7.9×10^{-1}	11
Uranium-238+D	2.7×10^{0}	77	2.0×10^{1}	84	1.8×10^{0}	69	1.6×10^{0}	66	3.1×10^{0}	44
Technetium-99	2.3×10^{-2}	1	6.0×10^{-2}	<1	7.4×10^{-3}	<1	2.1×10^{-12}	<1	3.1×10^{0}	45
Iodine-129	4.8×10^{-5}	<1	2.8×10^{-5}	<1	1.7×10^{-5}	<1	2.5×10^{-2}	1	1.8×10^{-5}	<1
TOTAL	4×10^{0}		2×10^{1}		3×10^{0}		2×10^{0}		7×10^{0}	

5.0 cm/yr Recharge Rate (maximum exposure 600 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	8.9×10^{-1}	25	4.4×10^{0}	18	8.9×10^{-1}	33	8.9×10^{-1}	36	8.9×10^{-1}	12
Uranium-238+D	2.7×10^{0}	75	2.0×10^{1}	82	1.8×10^{0}	66	1.6×10^{0}	63	3.1×10^{0}	43
Technetium-99	2.3×10^{-2}	<1	6.1×10^{-2}	<1	7.5×10^{-4}	<1	2.1×10^{-12}	<1	3.2×10^{0}	36
Iodine-129	4.8×10^{-5}	<1	2.8×10^{-5}	<1	1.7×10^{-7}	<1	2.5×10^{-2}	1	1.8×10^{-5}	<1
TOTAL	4×10^{0}		2×10^{1}		3×10^{0}		2×10^{0}		7×10^{0}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

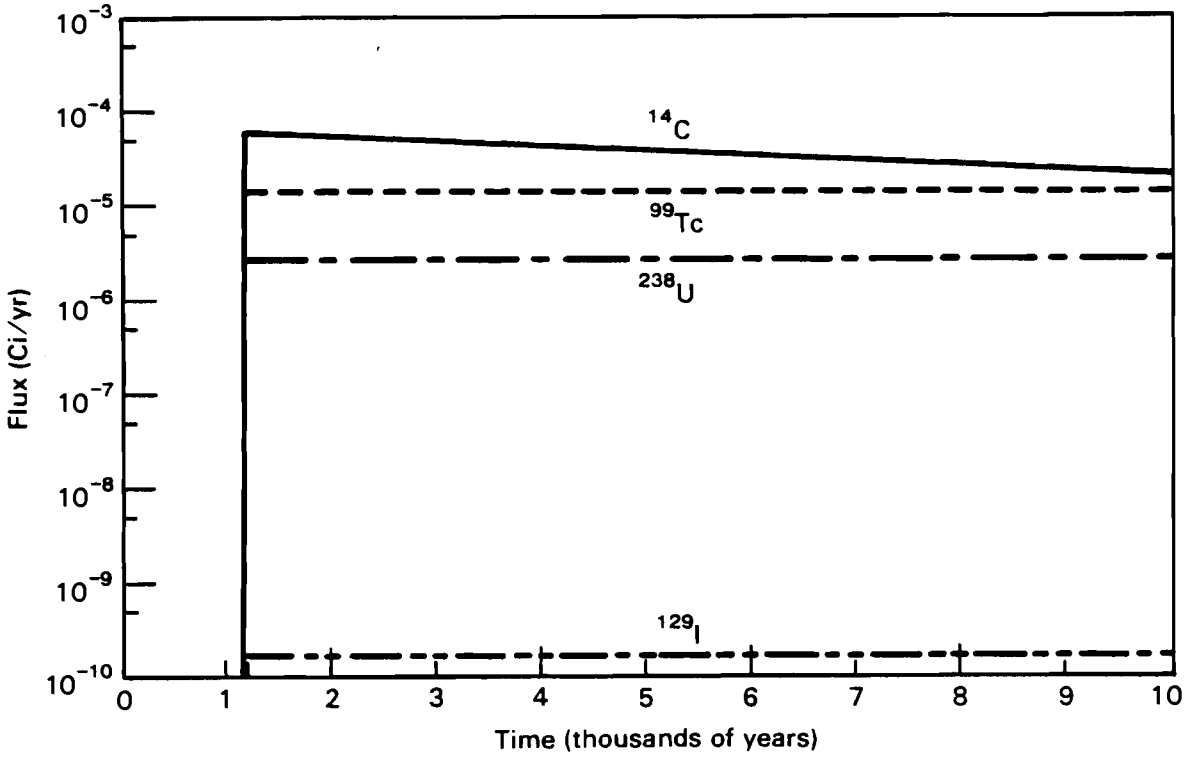


FIGURE 6.4. 10,000-year Release to Columbia River, 0.5 cm/year Recharge Rate

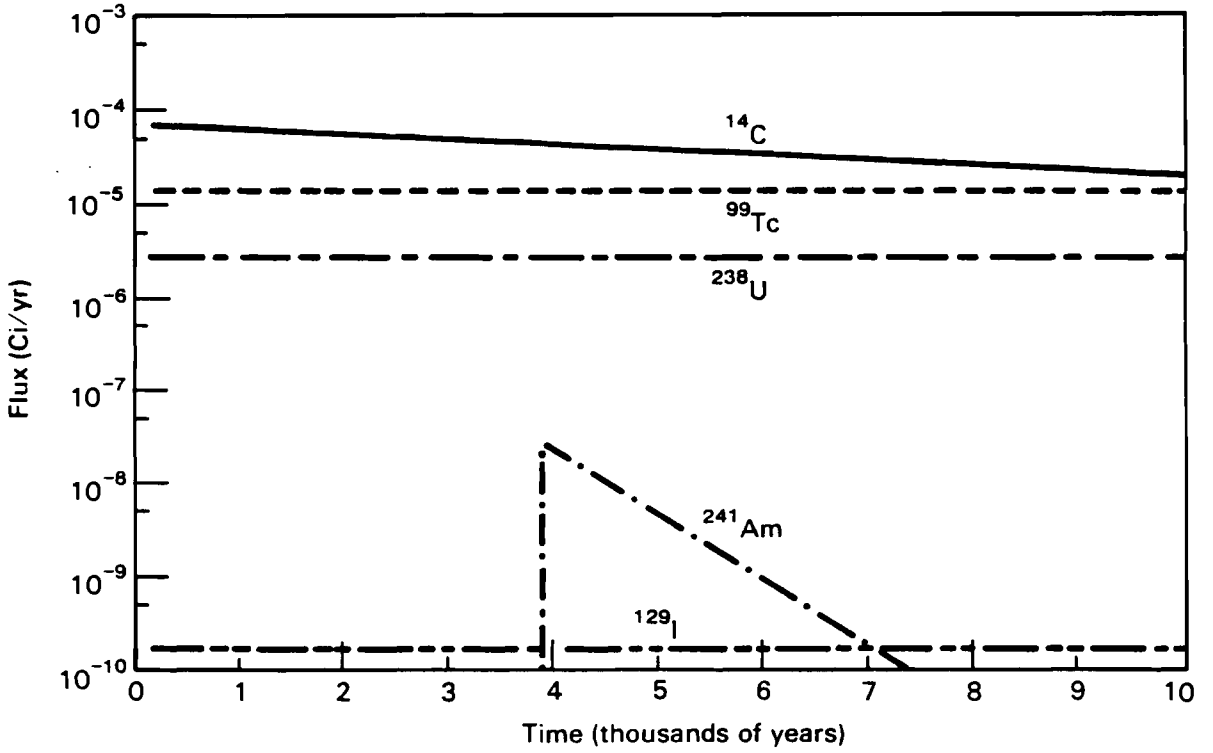


FIGURE 6.5. 10,000-year Release to Columbia River, 5.0 cm/year Recharge Rate

125,000 people swim or boat in the river (McCormack, Ramsdell and Napier 1984). It is assumed that a maximum of 5 million affected individuals will live at any point in time downstream along the Columbia River between Hanford and the river's mouth during the next 10,000 years. For this many people to be affected, a very large increase in the amount of irrigated land in both Washington and Oregon would be required, concurrent with a large increase in overall population. Thus, about 410 million people are assumed to live along the Columbia River over the 10,000-year period. The total dose that a group this size would receive from naturally occurring background sources is nearly 3 billion person-rem.

People who live along the Columbia River downstream of Hanford could be subject to doses resulting from the release of radionuclides from the grout. The gradual release of contaminants to the river would cause a slow increase in dose rate to a peak, followed by a gradual decline. There could be more than one peak, separated in time from the others, caused by the different mobilities of the radionuclides released from grout. The total dose to all people living over the next 10,000 years depends mostly on the total activity of each radionuclide (curies) released, but the rate of release controls the dose rate to any one individual.

Tables 6.7 and 6.8 show the doses that people living downriver of the Hanford Site would receive from PSW grout. Ten-thousand-year population doses are given in Table 6.9. As in the previous scenarios, the dominant radionuclide is uranium-238.

6.4 INTRUDER IMPACTS

In accordance with NRC guidelines (U.S. NRC 1981), active institutional controls cannot be relied upon for environmental protection for more than 100 years after disposal. After that time, passive institutional controls, such as markers, monuments, and public records, are the only mechanisms to inhibit intrusion into the waste. For this PA, intrusion events were analyzed at 100 years when active institutional control is assumed to cease at the Hanford Site. In addition, intrusion was also postulated at 400, 1,000, and 10,000 years after disposal. Radiological impacts were estimated at these

TABLE 6.7. Calculated Potential Maximum Radiation Doses to Individuals from Radionuclide Migration to the Columbia River, mrem (Annual Doses)

0.5 cm/yr Recharge Rate (maximum exposure 3000 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	4.1×10^{-10}	9	2.0×10^{-9}	7	4.1×10^{-10}	12	4.1×10^{-10}	14	4.1×10^{-10}	4
Uranium-238+D	4.0×10^{-9}	90	2.6×10^{-8}	93	3.0×10^{-9}	88	2.6×10^{-9}	85	4.5×10^{-9}	47
Technetium-99	3.4×10^{-11}	1	8.9×10^{-11}	<1	1.1×10^{-11}	<1	2.8×10^{-16}	<1	4.6×10^{-9}	49
Iodine-129	8.0×10^{-14}	<1	4.7×10^{-14}	<1	2.8×10^{-14}	<1	4.1×10^{-11}	1	3.1×10^{-14}	<1
TOTAL	4×10^{-9}		3×10^{-8}		3×10^{-9}		3×10^{-9}		1×10^{-8}	

5.0 cm/yr Recharge Rate (maximum exposure 2000 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	4.1×10^{-10}	9	2.0×10^{-9}	7	4.1×10^{-10}	12	4.1×10^{-10}	14	4.1×10^{-10}	4
Uranium-238+D	4.0×10^{-9}	90	2.6×10^{-8}	93	3.0×10^{-9}	88	2.6×10^{-9}	85	4.5×10^{-9}	47
Technetium-99	3.4×10^{-11}	1	8.9×10^{-11}	<1	1.1×10^{-11}	<1	2.8×10^{-16}	<1	4.6×10^{-9}	49
Iodine-129	8.0×10^{-14}	<1	4.7×10^{-14}	<1	2.8×10^{-14}	<1	4.1×10^{-11}	1	3.1×10^{-14}	<1
Americium-241	2.6×10^{-14}	<1	7.7×10^{-14}	<1	4.3×10^{-14}	<1	2.4×10^{-14}	<1	2.6×10^{-14}	<1
TOTAL	4×10^{-9}		3×10^{-8}		3×10^{-9}		3×10^{-9}		1×10^{-8}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

TABLE 6.8. Calculated Potential Maximum Radiation Doses to Individuals from **Radionuclide Migration to the Columbia River**, mrem (Lifetime Doses)

0.5 cm/yr Recharge Rate (maximum exposure 3000 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	2.9×10^{-8}	9	1.4×10^{-7}	7	2.9×10^{-8}	12	2.9×10^{-8}	14	2.9×10^{-8}	4
Uranium-238+D	2.8×10^{-7}	90	1.8×10^{-6}	93	2.1×10^{-7}	88	1.8×10^{-7}	85	3.2×10^{-7}	47
Technetium-99	2.4×10^{-9}	1	6.2×10^{-9}	<1	7.6×10^{-10}	<1	2.0×10^{-14}	<1	3.2×10^{-7}	49
Iodine-129	5.6×10^{-12}	<1	3.3×10^{-12}	<1	1.9×10^{-12}	<1	2.9×10^{-9}	1	2.1×10^{-12}	<1
TOTAL	3×10^{-7}		2×10^{-6}		2×10^{-7}		2×10^{-7}		7×10^{-7}	

5.0 cm/yr Recharge Rate (maximum exposure 2000 years after disposal)

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[a]	Percent Contribution
Carbon-14	2.9×10^{-8}	9	1.4×10^{-7}	7	2.9×10^{-8}	12	2.9×10^{-8}	14	2.9×10^{-8}	4
Uranium-238+D	2.8×10^{-7}	90	1.8×10^{-6}	93	2.1×10^{-7}	88	1.8×10^{-7}	85	3.2×10^{-7}	47
Technetium-99	2.4×10^{-9}	1	6.2×10^{-9}	<1	7.6×10^{-10}	<1	2.0×10^{-14}	<1	3.2×10^{-7}	49
Iodine-129	5.6×10^{-12}	0	3.3×10^{-12}	<1	1.9×10^{-12}	<1	2.9×10^{-9}	1	2.1×10^{-12}	<1
Americium-241	1.8×10^{-12}	<1	5.4×10^{-12}	<1	3.0×10^{-12}	<1	1.7×10^{-12}	<1	1.8×10^{-12}	<1
TOTAL	3×10^{-7}		2×10^{-6}		2×10^{-7}		2×10^{-7}		7×10^{-7}	

(a) GI-LLI = gastrointestinal tract - lower large intestine.

TABLE 6.9. 10,000-year Cumulative Population Doses from Radionuclide Migration to the Columbia River, person-rem[a]

0.5 cm/yr Recharge Rate

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[b]	Percent Contribution
Carbon-14	1.8×10^{-2}	15	9.0×10^{-2}	11	1.8×10^{-2}	17	1.8×10^{-2}	20	1.8×10^{-2}	8
Uranium-238+D	1.0×10^{-1}	84	7.0×10^{-1}	88	8.5×10^{-2}	82	7.2×10^{-2}	79	1.0×10^{-1}	46
Technetium-99	9.6×10^{-4}	1	2.5×10^{-3}	<1	3.1×10^{-4}	<1	7.9×10^{-9}	<1	1.0×10^{-1}	46
Iodine-129	2.2×10^{-6}	<1	1.3×10^{-6}	<1	7.7×10^{-7}	<1	1.3×10^{-3}	1	8.5×10^{-7}	<1
TOTAL	1×10^{-1}		8×10^{-1}		1×10^{-1}		9×10^{-2}		2×10^{-1}	

5.0 cm/yr Recharge Rate

Radionuclide	Total Body	Percent Contribution	Bone	Percent Contribution	Lung	Percent Contribution	Thyroid	Percent Contribution	GI-LLI[b]	Percent Contribution
Carbon-14	1.9×10^{-2}	16	9.5×10^{-2}	11	1.9×10^{-2}	18	1.9×10^{-2}	20	1.9×10^{-2}	9
Uranium-238+D	1.0×10^{-1}	83	8.0×10^{-1}	89	8.7×10^{-2}	82	7.4×10^{-2}	78	1.0×10^{-1}	46
Technetium-99	9.9×10^{-4}	1	2.6×10^{-3}	<1	3.2×10^{-4}	<1	8.1×10^{-9}	<1	1.0×10^{-1}	46
Iodine-129	2.3×10^{-6}	<1	1.3×10^{-6}	<1	7.7×10^{-7}	<1	1.2×10^{-3}	1	8.5×10^{-7}	<1
Americium-241	1.5×10^{-4}	<1	6.7×10^{-4}	<1	2.2×10^{-4}	<1	1.3×10^{-4}	<1	1.6×10^{-4}	<1
TOTAL	1×10^{-1}		9×10^{-1}		1×10^{-1}		9×10^{-2}		2×10^{-1}	

(a) Cumulative population dose received by the local population over 10,000 years with an assumed 70-yr individual lifetime.
(b) GI-LLI = gastrointestinal tract - lower large intestine.

times of intrusion and are described below. Radionuclide concentrations used in the following scenarios were presented in Table 4.4.

6.4.1 Drilling

Drilling, either for water wells or for mineral exploration, is a potential mechanism for moving buried waste directly to the earth's surface with little indication that the waste has been encountered. Monuments, barriers, and markers may reduce the likelihood of drilling, but they cannot preclude it.

In the drilling scenario, a well 30 cm in diameter is bored through the grout. The grout vault is 7 m thick; hence, 0.5 m^3 of grout is brought to the surface. Drilling through the waste form itself is assumed to take 1 hour. During this time, the driller breathes contaminated dust with a mass loading of 1 x 10^{-4} g/m^3 of air. The drill tailings are assumed to be spread over a 100-m^2 area.

The drillers are assumed to spend 40 hours working in the immediate vicinity of the tailings. (The maximum annual dose includes that from external radiation received during drilling, plus the longer-term dose that results from inhalation of radionuclides in the contaminated dust.)

Maximum annual total-body radiation doses to members of the drill crew are presented in Table 6.10. The doses are dominated by external exposure to cesium-137 at early times and after disposal, uranium-238 at longer time periods.

6.4.2 Excavation

Several plausible excavation events that involve major disturbances of the grout site can be postulated. These include construction projects required for highway or canal building, or, on a smaller scale, for residential basements. In these cases, workers operating heavy machinery can be assumed to be in a "hole in the ground," essentially surrounded by contaminated soil. The hole could range from relatively small (for a basement) to quite large (for a canal), but the direct exposure source and the contaminated dust concentration would be about the same in either case. The workers in the

TABLE 6.10. Calculated Maximum Radiation Doses to Individuals Resulting from the Drilling Scenario (Annual Doses)

Time, yr	Organ	Individual Maximum Annual Dose, mrem/yr	Dominant Pathway	Dominant Nuclide
100	Total Body	4×10^{-1}	External	^{137}Cs
	Bone	4×10^{-1}	External	^{137}Cs
	Lung	4×10^{-1}	External	^{137}Cs
	Thyroid	4×10^{-1}	External	^{137}Cs
	LLI(a)	4×10^{-1}	External	^{137}Cs
400	Total Body	4×10^{-4}	External	^{137}Cs
	Bone	4×10^{-4}	External	^{137}Cs
	Lung	4×10^{-4}	External	^{137}Cs
	Thyroid	4×10^{-4}	External	^{137}Cs
	LLI(a)	4×10^{-4}	External	^{137}Cs
1000	Total Body	3×10^{-6}	External	^{238}U
	Bone	5×10^{-6}	External	^{238}U
	Lung	2×10^{-5}	Inhalation	^{239}Pu
	Thyroid	3×10^{-6}	External	^{238}U
	LLI(a)	3×10^{-6}	External	^{238}U
10,000	Total Body	2×10^{-6}	External	^{238}U
	Bone	4×10^{-6}	External	^{238}U
	Lung	1×10^{-5}	Inhalation	^{239}Pu
	Thyroid	2×10^{-6}	External	^{238}U
	LLI(a)	2×10^{-6}	External	^{238}U

(a) LLI = lower large intestine.

hole would be exposed to direct radiation from radionuclides in the soil and to resuspended dust from the construction activity.

An individual operating heavy equipment is assumed to work in the contaminated area for 80 hours. A contaminated dust loading of 1×10^{-2} g/m^3 of

air is assumed. The excavated waste is assumed to be uniformly mixed with soil that has a density of 1.7×10^6 g/m^3.

Maximum annual whole-body radiation doses to workers who excavate into the grout at various future times are presented in Table 6.11. Again, maximum annual total-body radiation doses are dominated by external exposures from cesium-137 and uranium-238.

6.4.3 Residential Home Garden

The resettlement of the Hanford Site is assumed to occur following the cessation of active institutional control of the Site. It is believed that hypothetical resettlement would occur first along parts of the Hanford Site relatively close to the Columbia River because of the availability of water from both the river and groundwater at shallow depths. However, for the sake of conservatism, potential future occupancy is also assumed near or at locations of disposed waste. For waste sites in the 200-Area plateau (where PSW grout is planned to be disposed), this type of resettlement is believed to be applicable only to a few individuals, rather than a systematic settlement.

Without active institutional controls, and with disregard of passive institutional controls such as permanent markers and public records, waste disposal areas could possibly be used for residential purposes. People could build homes and conduct routine subsistence activities over buried waste sites. Food crops, for either domestic or animal consumption, could be grown. (Individuals in this scenario are assumed to use uncontaminated well water. For impacts directly resulting from use of contaminated well water, see Section 6.3.2.) The resident would consequently be exposed to low levels of direct radiation from the buried grout and also to radionuclides via ingestion of contaminated crops grown in the site. Crop contamination is a function of the depth of the grout, the ability of the grout vault to prevent root intrusion, and the overall surface area used for gardening.

Exposure pathways for the residential home garden scenario are ingestion of contaminated food and external radiation.

TABLE 6.11. Calculated Maximum Radiation Doses to Individuals Resulting from the Excavation Scenario (Annual Doses)

Time, yr	Organ	Individual Maximum Annual Dose, mrem/yr	Dominant Pathway	Dominant Nuclide
100	Total Body	4×10^{1}	External	^{137}Cs
	Bone	4×10^{1}	External	^{137}Cs
	Lung	4×10^{1}	External	^{137}Cs
	Thyroid	4×10^{1}	External	^{137}Cs
	LLI(a)	4×10^{1}	External	^{137}Cs
400	Total Body	4×10^{-2}	External	^{137}Cs
	Bone	5×10^{-2}	External	^{137}Cs
	Lung	8×10^{-2}	Inhalation	^{239}Pu
	Thyroid	4×10^{-2}	External	^{137}Cs
	LLI(a)	4×10^{-2}	External	^{137}Cs
1000	Total Body	7×10^{-4}	Inhalation	^{239}Pu
	Bone	2×10^{-2}	Inhalation	^{239}Pu
	Lung	6×10^{-2}	Inhalation	^{239}Pu
	Thyroid	3×10^{-4}	External	^{238}U
	LLI(a)	3×10^{-4}	External	^{238}U
10,000	Total Body	5×10^{-4}	Inhalation	^{239}Pu
	Bone	1×10^{-2}	Inhalation	^{239}Pu
	Lung	5×10^{-2}	Inhalation	^{239}Pu
	Thyroid	2×10^{-4}	External	^{238}U
	LLI(a)	2×10^{-4}	External	^{238}U

(a) LLI = lower large intestine.

Dietary parameters represent the vegetable diet of the "Hanford maximum individual" (McCormack, Ramsdell and Napier 1984). The individual in this scenario is assumed to grow 25% of his diet in the contaminated soil. The garden is postulated to be located directly above a PSW grout vault.

One percent of the crop roots are in contact with the waste. For external radiation, the individual is exposed to the contaminated soil for 8,770 hours per year.

Calculated maximum annual whole-body radiation doses to individuals resulting from the residential home garden scenario are listed in Table 6.12. Individual maximum annual radiation doses are dominated by the ingestion pathway. Strontium-90 and cesium-137 control the doses for 400 years, while carbon-14 and technetium-99 control the doses after 1,000 years.

6.4.4 Post-Drilling Habitation

The doses to people who directly contact the buried wastes represent only a portion of the impact of intrusion into the grout site. Drilling results in waste being physically disturbed and distributed in the local environment. This contamination could represent a source of radiation exposure to people living on or near the site of the original disturbance long after the original redistribution. As in the example of the residential home garden scenario, people who live on or near the waste would be exposed to direct radiation from contamination in the soil, and to ingestion of garden foods grown in the contaminated soil.

In the post-drilling habitation scenario, waste brought to the surface by the drilling scenario is assumed to be further distributed throughout a 15-cm plow layer in a garden that is 2,500 square meters in area. Twenty-five percent of the individual's vegetable intake is assumed to come from this garden. The individual is assumed to spend 2,000 hours per year outside, exposed to resuspended dust and to surface contamination.

Calculated maximum annual doses to individuals living on the grout site are presented in Table 6.13. Total-body doses are dominated by strontium-90 ingestion at 400 years, carbon-14 ingestion at 1,000 years, and inhalation of plutonium-239 at 10,000 years.

6.4.5 Post-Excavation Habitation

The post-excavation habitation scenario follows directly from the excavation activities described in Section 6.4.2. As in the post-drilling scenario,

TABLE 6.12. Calculated Maximum Radiation Doses to Individuals Resulting from the **Residential Home Garden Scenario** (Annual Doses)

Time, yr	Organ	Individual Maximum Annual Dose, mrem/yr	Dominant Pathway	Dominant Nuclide
100	Total Body	6×10^{1}	Ingestion	^{90}Sr
	Bone	2×10^{2}	Ingestion	^{90}Sr
	Lung	2×10^{-2}	Ingestion	^{137}Cs
	Thyroid	2×10^{-6}	Ingestion	^{129}I
	LLI[(a)]	7×10^{-0}	Ingestion	^{90}Sr
400	Total Body	6×10^{-2}	Ingestion	^{90}Sr
	Bone	2×10^{-1}	Ingestion	^{90}Sr
	Lung	3×10^{-5}	Ingestion	^{137}Cs
	Thyroid	2×10^{-6}	Ingestion	^{129}I
	LLI[(a)]	2×10^{-2}	Ingestion	^{14}C
1000	Total Body	1×10^{-2}	Ingestion	^{14}C
	Bone	7×10^{-2}	Ingestion	^{14}C
	Lung	4×10^{-6}	Ingestion	^{99}Tc
	Thyroid	2×10^{-6}	Ingestion	^{129}I
	LLI[(a)]	1×10^{-2}	Ingestion	^{14}C
10,000	Total Body	4×10^{-3}	Ingestion	^{14}C
	Bone	3×10^{-2}	Ingestion	^{14}C
	Lung	4×10^{-6}	Ingestion	^{99}Tc
	Thyroid	2×10^{-6}	Ingestion	^{129}I
	LLI[(a)]	5×10^{-3}	Ingestion	^{14}C

(a) LLI = lower large intestine.

people were postulated to live in a home constructed at the site and to consume food grown in an adjacent small garden. However, it was considered to be extremely unlikely that excavation activities would bring materials to the

TABLE 6.13. Calculated Maximum Radiation Doses to Individuals Resulting from the Post-Drilling Scenario (Annual Doses)

Time, yr	Organ	Individual Maximum Annual Dose, mrem/yr	Dominant Pathway	Dominant Nuclide
100	Total Body	1×10^{1}	Ingestion	^{90}Sr
	Bone	4×10^{1}	Ingestion	^{90}Sr
	Lung	2×10^{0}	External	^{137}Cs
	Thyroid	2×10^{0}	External	^{137}Cs
	LLI[(a)]	3×10^{0}	External	^{137}Cs
400	Total Body	1×10^{-2}	Ingestion	^{90}Sr
	Bone	6×10^{-2}	Ingestion	^{90}Sr
	Lung	1×10^{-2}	Inhalation	^{239}Pu
	Thyroid	2×10^{-3}	External	^{137}Cs
	LLI[(a)]	5×10^{-3}	Ingestion	^{14}C
1000	Total Body	4×10^{-3}	Ingestion	^{14}C
	Bone	4×10^{-2}	Inhalation	^{239}Pu
	Lung	9×10^{-3}	Inhalation	^{239}Pu
	Thyroid	1×10^{-5}	External	^{239}Pu
	LLI[(a)]	2×10^{-6}	Ingestion	^{14}C
10,000	Total Body	2×10^{-3}	Inhalation	^{239}Pu
	Bone	3×10^{-2}	Inhalation	^{239}Pu
	Lung	7×10^{-3}	Inhalation	^{239}Pu
	Thyroid	9×10^{-6}	External	^{238}U
	LLI[(a)]	9×10^{-4}	Ingestion	^{14}C

(a) LLI = lower large intestine.

surface for redistribution from depths greater than 5 m, and therefore, no impacts were projected to be associated with the reference 5-m depth of soil overburden.

6.5 SUMMARY OF RADIOLOGICAL IMPACTS

Table 6.14 summarizes the long-term radiological impacts associated with the disposal of grouted PSW at Hanford. The maximum dose calculated for each type of scenario is shown. The major types of scenarios represent impacts from drinking water at a downgradient well, using a full garden at this same well, using water from the downgradient river, and intruding directly into the grouted waste. The summary doses are compared to various radiological standards as a means of measuring the effectiveness of the grout disposal system. Correspondent regulations are also listed for comparison.

TABLE 6.14. Summary of Maximum Radiological Impacts

Scenario	Dose, mrem/yr Total Body/ Critical Organ	Regulatory Dose Limit, mrem/yr Total Body/Critical Organ	Dominant Radionuclide
Drinking Water, 0.5 cm/yr recharge	0.02/0.3	4/4[a]	^{238}U
Full Garden, 5.0 cm/yr recharge	0.05/0.4	25/75[b]	^{238}U
River, both recharges	$4 \times 10^{-9}/3 \times 10^{-8}$	25/75[b]	^{238}U
Intruder (residential home garden)	60/200	500/1500[b,c]	^{90}Sr, ^{137}Cs

(a) WAC (1985).
(b) U.S. NRC (1982a).
(c) U.S. DOE (1981a).

6.5.1 Regulatory Review

The current body of radiological regulations and guidelines pertaining to the protection of public health and the environment were reviewed to provide a way to measure the long-term performance of the PSW grout system. Included in this review, for direct applicability and indirect guidance, were DOE orders and federal and state regulations, including those promulgated by the EPA, the NRC, and the State of Washington.

The regulatory review indicated that regulations pertaining to the permanent disposal of both DOE and commercial LLW are currently developing and changing. In 1983, the EPA published an advanced notice of public rulemaking for LLW disposal (U.S. EPA 1983). These regulations are being developed and

are expected to be promulgated as LLW standards in the Code of Federal Regulations as 40 CFR 193. After promulgation, these regulations would be expected to apply to the disposal of future PSW grout; however, the schedule for final issuance is currently unknown.

Although the DOE is not legally bound to comply with NRC requirements, it is useful to inspect performance criteria that have been established for land disposal of commercial LLW, particularly because they are the result of a lengthy environmental impact study that included the incorporation of public comments (U.S. NRC 1982b). The NRC issued regulations for disposal of commercial radioactive wastes in shallow-land facilities as contained in 10 CFR 61 (U.S. NRC 1982a). The regulations established procedures, terms, criteria and conditions upon which the NRC will issue a license for the disposal of wastes containing byproduct, source, and special nuclear materials.

Requirements to protect the general population from releases of radioactivity are established in 10 CFR 61.41 (U.S. NRC 1982a). According to these requirements, concentrations of radionuclides transported through all pathways, including groundwater, surface water, air, soil, plants, or animals must not result in an annual dose to an individual exceeding

- 25 mrem to the whole body
- 75 mrem to the thyroid
- 25 mrem to any other organ.

Additionally, the regulation stipulated that "design, operation, and closure of the land-disposal facility must ensure protection of any individual inadvertently intruding into the disposal site and occupying the site or contacting the waste at any time after active institutional controls over the disposal site are removed" (U.S. NRC 1982a).

The NRC approach to ensuring the long-term protection of the inadvertent intruder was to establish maximum waste concentrations for a series of disposal systems of increasing stability and expected long-term performance. To determine the limits, the NRC evaluated the expected performance of each disposal system, using several hypothetical exposure scenarios, and using an annual dose limit of 500 mrem to the whole body or bone and 1500 mrem/yr to

any other critical organ. The inadvertent intruder scenarios evaluated in our performance assessment closely parallel the NRC scenarios; therefore, the impacts of inadvertently intruding into the PSW disposal site are "benchmarked" against the design criteria of 500/1500 mrem/yr.

In lieu of specific EPA and NRC regulations pertaining to permanent disposal of DOE LLW, other regulations were examined. In particular, the following DOE orders, as applied during the operational phase of disposal, were reviewed for guidance:

- Order 5480.2, "Radioactive Waste Management" (U.S. DOE 1984)
- Order 5480.1A, "Environmental Protection, Safety and Health Protection Programs for DOE Operations" (U.S. DOE 1981a).

Regulations specific to the management of radioactive waste are contained in DOE Order 5820.2, "Radioactive Waste Mangement" (U.S. DOE 1984). Chapter III of DOE Order 5820.2, "Management of Low-Level Waste," established DOE policy and guidelines for management and disposal of LLW. This order established the policy that LLW generated by DOE operations shall be disposed ". . . where practical, by shallow land burial or greater confinement disposal. Site-specific requirements for waste acceptance and disposal, site selection, site design, site operation, and site closure/post-closure shall be developed and implemented by field organizations . . .". No quantitative performance objectives are included in the order; however, additional guidance is being prepared.

The DOE Order 5480.1A, "Environmental Protection, Safety and Health Protection Programs for DOE Operations," established an overall framework of program requirements for safety and environmental and health protection (U.S. DOE 1981a). The order set forth radiation protection standards for operating DOE facilities. Chapter XI of DOE Order 5480.1A includes the dose limits for members of the general public in unrestricted areas in the vicinity of DOE facilities. The exposure limits were maximum annual dose equivalents of 500 mrem to the whole body, gonads, or bone marrow, and 1500 mrem to any other organ. Calculation of compliance with these limits must include an analysis of all potential pathways including groundwater, air, and direct exposure as well as consumption of contaminated foodstuffs (U.S. DOE 1981b).

These standards apply to routine DOE operations, defined as normal planned operations, and do not include actual or potential accidents or unplanned releases.

The DOE is in the process of revising its radiation standards for protection of the public in the vicinity of DOE facilities. In 1985, DOE issued guidance supplanting the dose limits outlined above with new dose limits based on the latest concepts and dosimetry of the International Commission on Radiological Protection (ICRP 1977, 1979). Interim standards, effective July 1, 1985, limit the continuous dose to any member of the public to 100 mrem/yr from all routine DOE operations at a DOE site.

This dose must be calculated using the dosimetry models of ICRP 26/30. As discussed in Chapter 6.0, Section 6.2.1, the capability to calculate long-term impacts using the newer dosimetry is being developed at this time. However, because the doses presented in this analysis are calculated using ICRP 2 dosimetry, they must be compared to dose limits that are based on ICRP 2 models.

Finally, federal and state drinking water standards were reviewed for guidance concerning the radiological protection of the groundwater. The Safe Drinking Water Act (SDWA) was enacted in 1974 to protect the nation's drinking water. Under the authority of SDWA, the EPA issued National Interim Primary Drinking Water Regulations in 40 CFR 141 (U.S. EPA 1984). The 40 CFR 141 regulations set forth radiological standards. The State of Washington has adopted the EPA standards as drinking water limits in WAC 248-54 (WAC 1985). The radionuclide standards for Washington State drinking water supply systems are established such that "the maximum contaminant level for beta particle and photon radioactivity from man-made radionuclides is that the average annual concentration shall not produce an annual dose equivalent to the total body or any internal organ greater than four millirem/year" (WAC 1985).

6.5.2 Comparison to Regulations

The long-term radiological impacts resulting from migration of radionuclides in groundwater are dominated by uranium-238. At 100 years after disposal of PSW, the intruder impacts are dominated by strontium-90 and cesium-137; after 400 years, the impacts are approximately 1000 times less.

The calculated release of radionuclides from PSW grout to the Columbia River results in extremely low incremental increases of radioactivity. The 10,000-year cumulative impact from PSW is less than 1 person-rem. This can be compared to the dose from natural background radiation that the downriver population would receive over 10,000 years: 3 billion person-rem.

7.0 GLOSSARY

active institutional control - in this document, active institutional control means continued federal control of the Hanford Site along with maintenance and surveillance of facilities and waste sites

advection - the transport of a substance solely by the bulk phase movement of a fluid such as water

aquifer - an underground bed or stratum of earth, gravel, or porous stone that contains water. The water can be pumped to the surface through a well or it can emerge naturally as a spring

barrier - any medium that retards the movement of emplaced radioactive or nonradioactive material or reduces the probability of human access to the material. (Examples are engineered features, including waste containers, waste form, or backfill material; a natural geologic medium; or institutional site access and use restrictions.)

conservative - conservative choices of parameters or assumptions are those that would tend to overestimate rather than underestimate impacts

constant release rate - a rate of release (amount released/time) that does not vary over time

diffusion-controlled release rate - a rate of release (amount released/time) dependent on the rate of diffusion of the released substance from inside the source to the surface of the source

disposal system - combination of waste preparation steps, engineered and natural barriers, performance verification methods, and in situ marking systems that contain and isolate waste after disposal

distribution coefficient (K_d) - the ratio of the concentration of a solute sorbed by solids in a porous media to the concentration of solute remaining in solution

dose - the term "dose" is used throughout this report as a shorthand notation where the term dose-equivalent (calculated in mrem) is intended

dose equivalent - the product of absorbed dose, quality factor, distribution factor, and other modifying factors necessary to evaluate the effects of irradiation received by exposed persons, so that the different characteristics of the exposure are taken into account

effective dose equivalent - the sum of the products of the dose equivalent to individual organs and tissues and appropriate weighing factors representing the risk of health relative to that for an equal dose to the whole body

evapotranspiration - the combined loss of water from soil by evaporation and by transpiration from the surfaces of plant structures

exposure - the condition of being made subject to the action of radiation; a measure, in roentgens, of the ionization produced in air by x-ray or gamma radiation

fingering - the formation and downward propagation of (finger-shaped) regions of preferential flow caused by viscosity differences between fluids occupying the pore space of a porous solid

flux - quantity of substance moving past a unit area (perpendicular to flow direction) per unit time

groundwater - water below the land surface in a zone of saturation

grout - a slurry mixture of cement, water, fly ash, and clay that sets up as a solid mass and is used for waste fixation or immobilization

half-life - the time required for a radionuclide's activity to decay to half its value, used as a measure of the persistence of radioactive materials; each radionuclide has a unique half-life

hazardous waste - a solid waste, or combination of solid wastes, which because of its quantity, concentration, or physical, chemical or infectious characteristics may:

a) cause, or significantly contribute to an increase in mortality or an increase in serious irreversible, or incapacitating reversible, illness; or

b) pose a substantial present or potential hazard to human health or the environmental when improperly treated, stored, transported, or disposed of, or otherwise managed.

hydraulic conductivity - the parameter relating the volumetric flux to the driving force in flow through a porous medium (particularly water through soil); a function of both the porous medium and the properties of the fluid

inadvertent intrusion - human activity, such as home excavation, resource mining, and well digging, that accidentally breaches a waste site

institutional control - see active institutional control or passive institutional control

intruder - a person who violates site and marker boundaries to disturb a waste site

isotope - one of several different species of a given chemical element; different isotopes are distinguished by different numbers of neutrons in the atomic nucleus

K_d - see distribution coefficient

leach - to dissolve out the soluble components of a solid by contact with water or other solvent

low-level waste (LLW) - radioactive waste not classified as high-level waste, transuranic waste, or spent nuclear fuel

model - a conceptual or mathematical representation of the chemical, physical, and biological behavior in the natural environment; the computational implementation of which often requires a computer code

parameters - constant or variable factors (mathematical variables) contained in a mathematical model which, when given different numerical values, will result in different values of some desired output variable

passive institutional control - period following closure and deactivation of a disposal site during which public access and use is restricted by continued government ownership and restrictions on either land or resource area by the use of markers to delineate boundaries and potential health hazards to intruders and the use of public records, archives, and other methods to preserve information about the location, design, and contents of the disposed system

pathway - the movement of materials from the source to locations of interest

performance assessment - an analysis that identifies the events and processes that might affect the waste disposal system, examines their effects upon its natural and engineered barriers, and estimates the probabilities and consequences of those events and processes

Phosphate/Sulfate Waste from N Reactor Operations - a combination of two liquid low-level waste streams generated by the N Reactor at the Hanford Site, comprised of one stream generated periodically during decontamination activities, another semi-continuously as a result of back-flushing ion exchange resins used to purify the water in the spent-fuel storage basin

piecewise constant - constant within a discrete piece of a region of interest; with different pieces of the region having different constant values

radionuclide - any nuclide that emits radiation

recharge - the net process of water entering a saturated aquifer at the water table after having percolated downward through the soil profile. Recharge is a fraction of the annual rainfall; the remainder is evaporated from the bare surface or transpired by plants.

sorb - to adsorb onto a solid surface or to become absorbed into the body of a solid material

sorption - a general term used to encompass the processes of physical and chemical absorption or adsorption, ion exchange, and dialysis

standards - a quantitative measure of criteria satisfaction, or prescriptive norms established to govern action, established from the perspective of criteria; can govern such things as radiation exposure, water quality, barrier and waste form design, and releases from a waste site

steady-state - constant with respect to time

streamlines - in a steady-state flow model, the paths that particles would move along if passively flowing along with the moving water; a line that is everywhere parallel to the direction of fluid flow at a given instant

stream tube - water flowing between two streamlines (two-dimensional); an imaginary tube whose wall is generated by streamlines passing through a closed curve

transmissivity - a coefficient relating the volumetric flow through a unit width of groundwater to the driving force (hydraulic potential); a function of the porous medium, fluid properties, and saturated thickness of the aquifer

unconfined aquifer - an aquifer that has a water table or surface at a atmospheric pressure

vadose zone - the unsaturated region of soil between the ground surface and the water table

vault - in the context of this report, a concrete disposal structure for containing grouted waste

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